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BACKGROUND STUDY ON THE DEVELOPMENT OF A STANDARD LEACHING TEST

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FOREWORD

When energy and material resources are extracted, processed, converted and used, the related pollutional impacts on our environment and even on our health often require that new and increasingly more efficient pollution control methods be used. The Industrial Environmental Research Laboratory-Cincinnati (IERL-Ci) assists in developing and demonstrating new and improved methodologies that will meet these needs both efficiently and economically.

The Office of Solid Waste has as its major goals, the improvement of solid waste management in order to protect public health and the environment, and the conservation of valuable material and energy resources. These objectives include, regulation of the management of hazardous wastes from the point of generation through disposal, regulation of the disposal on land of all other solid wastes, and establishment of resource recovery and conservation as the preferred solid waste management approach.

The research described in this report is a product of the efforts of the Industrial Environmental Research Laboratory-Cincinnati (IERL-Ci) and the Office of Solid Waste. The report deals with a study to develop a leaching test that can be used widely to assess the leaching characteristics of industrial wastes. The test procedure provides information regarding some materials that are likely to be leached from a waste, estimated release concentrations of these materials, and levels of release per unit weight of waste. The report suggests several criteria for discriminating between wastes that produce hazardous leachates and those that do not. It thereby provides data for decision makers of both government and industry alike contemplating residue leachate control from industrial sludge impoundment/municipal landfill co-disposal operations. Information on this subject beyond that supplied in the report may be obtained from the Hazardous Waste Management Division of the Office of Solid Waste, Washington, D. C. 20460 and the Oil and Hazardous Materials Spills Branch (IERL-Ci), Edison, New Jersey 08817.

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ABSTRACT

The principal objective of the research summarized in this report was to develop a leaching test which could be used widely to assess the leaching characteristics of industrial wastes. Detailed investigations were made regarding the best general type of test, and the test variables and operating conditions which must be standardized if the test is to be used by many laboratories and on different wastes.

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ABBREVIATIONS AND SYMBOLS

A. List of experimental test series and principal objectives of each

Series B—Biological effects
Series P—Elution time
Series PV—Agitation methods
Series R—Solid-liquid ratios: $\begin{cases} \text{R1—procedure R} \\ \text{R2—procedure C} \end{cases}$
Series V—Different landfill situations and solid-liquid ratios

B. List of wastes and designations used (see more complete list with sources in Chapter 5)

1. Shredded municipal refuse, City of Madison
2. Fly ash (unspecified further), power plant in Wisconsin
3. Papermill sludge (N or unspecified further), papermill in Wisconsin
4. Paint waste (AA), automobile assembly plant
5. Clarifier sludge (AA), automobile assembly plant
6. Fly ash (AA), automobile assembly plant
7. Fly ash (EPA), provided by EPA
8. Papermill sludge (EPA), provided by EPA
9. $CuO-Na_2SO_4$ slurry, provided by EPA
10. Wastewater treatment sludge, provided by EPA
11. Health and beauty care waste, provided by EPA (used in Figure 5 only, for this report)
12. Paint and ink waste, provided by EPA (used in Figure 6 only, for this report)
13. Oil/water waste, provided by EPA (used in Figure 7 only, for this report)

C. Other abbreviations

S.L. = Synthetic leachate modeled on municipal refuse landfill leachate
S.L.T. = Standard leaching test (i.e., of specified procedure)
ø = particle size as determined by sieving, mm.

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SECTION 1

INTRODUCTION

Recently an increased awareness of the potential for ground water pollution from industrial wastes disposed in landfills has become evident. Since many wastes will not produce polluting leachates when landfilled, there is need for criteria to discriminate between wastes that will produce hazardous leachates and those that will not. One such criterion is a short standardized leaching test which subjects the waste in the laboratory to simulated landfill conditions or to test conditions that can be related to landfill conditions. The test would evaluate the leaching potential of the waste under landfill conditions by indicating what constituents would leach out of the waste, how much of that material would leach out and under what conditions they will leach. This report summarizes a background study performed to develop such a leaching test.

SECTION 2

CONCLUSIONS

Subject to the scope and limitations of this study, the following conclusions have been reached.

1. Column tests are too time consuming and difficult to perform for a routine, widely used leaching test. A flask or batch test is preferred.
2. Solid-liquid separation is necessary prior to a leaching test in order to handle readily the wide variety of wastes and solids contents to be encountered.
3. Filtration with a 0.45 micron filter, along with appropriate filter aids, is adequate for solid-liquid separation prior to and between elutions of the recommended batch leaching test.
4. Because of the major differences in a waste's leaching characteristics as a result of the leaching media composition, no one media can give results adequate to describe properly the leaching characteristics of a waste. The use of distilled water to model mono-landfill conditions, municipal refuse landfill leachate to model co-disposal with municipal refuse, and appropriate special leachates to model co-disposal with other industrial wastes is recommended. These landfill possibilities represent the extremes which may occur in a municipal sanitary landfill, in which leachate composition may be controlled by the waste itself, actively decomposing municipal solid waste or another specific waste present in the landfill.
5. The leaching characteristics of an actively decomposing municipal refuse landfill leachate are greatly different from those of distilled water. A synthetic municipal refuse leachate has been developed which models the pH, redox potential, ionic strength, and complexing capability of actual leachate.
6. Information regarding both concentration and release of leached components from a waste is necessary for an understanding of the potential effect of landfilling that waste on water quality. No one leaching test procedure is adequate for providing both concentration and release information; therefore, separate batch procedures should be used, one providing information on maximum concentration, the other on maximum likely release of contaminants from a waste.

7. The operating conditions and test variables as evaluated and recommended were reasonable and adequate for the wastes tested (e.g., three elutions, 24 hours per elution, 1 to 10 dry weight waste to liquid volume ratio, etc.).

8. Proper interpretation of the results from the recommended procedure is critical to its usefulness. The test was designed to be aggressive; the numbers obtained are expected to be maximum values which will not be attained normally in an actual landfill.

SECTION 3

RECOMMENDATIONS

Many additional studies could be suggested as a result of this project, of which the following four are felt to be of particular importance.

1. The recommended leaching procedure should be used on a wider variety of wastes than was possible in the present study, and revised if warranted.
2. The procedure should be used on identical wastes by different laboratories to check on reproducibility from laboratory to laboratory.
3. There is a lack of data regarding leachate generation at full-scale industrial waste landfills with which results from the laboratory leaching procedure can be compared. Field verification studies are needed in which unattenuated and indiluted leachate from specific industrial wastes in mono- as well as co-landfill situations can be compared with appropriate leaching test results, preferably on a long term basis.
4. Additional methods of interpretation of leaching test results should be sought and refined as new data (especially from field verification work as in point (3)) become available.

SECTION 4

BASIC CONCEPTS

This chapter will discuss the philosophy of leaching tests and the test variables affecting the leaching test, review some of the leaching tests that have been developed, and discuss briefly the procedure which evolved from this work.

Intensive vs. Quick Tests

Two general approaches can be used to evaluate the leachability of waste material: (1) a very intensive study of waste leaching characteristics, or (2) a quick test using standardized procedures. The intensive study gives more information about the leaching characterists of a waste. Test conditions can be varied as needed, and the effects of different variables on the leaching characteristics can be studied. Such a test takes considerable time, money and personnel. The standardized test uses only predetermined testing conditions, and so cannot show the effects of the different variables of the waste leaching pattern. It can, however, give useful information in a short time which when properly interpreted can give an indication of the leaching characteristics of a waste. It is much cheaper, faster, and simpler than the intensive study.

Wastes generated in large quantities should be subjected to the intensive study, particularly wastes that are generated at different sites throughout the country but are of relatively uniform composition, e.g., fly ash or scrubber sludge. However, for the many wastes that are produced in relatively small amounts, a standard leaching test is more appropriate. The small amount of waste produced does not justify the expense involved in the intensive study unless the waste is of particular concern because of its characteristics or landfilling situation.

In an intensive test, factors affecting the test results will be varied and their effects observed and analyzed. In the standard test these factors will be set beforehand and their effects on the test results will not be observed. It is important to be aware of the test factors which are set in the standard test and their potential effect on the test results so that a proper interpretation can be made of the results.

Ideal and Practical Leaching Tests

Ideally a leaching test would determine four characteristics regarding the release of a parameter, A, from a waste:

1. the highest concentration of A to be found in the leachate;
2. the factors controlling this concentration,
3. the total amount of A available from a give amount of waste, and
4. the release pattern of A with time.

The last characteristic includes the kinetics of the release, physical or chemical changes occurring in the waste as it is leached, any effects of these changes on the release of A, and the influence of the waste on the leachate. Water quality standards are given in terms of concentration and since many toxic effects are concentration dependent as are most chemical reactions, concentration is of obvious interest. Maximum release is important when predicting the total amounts of A that may be leached from the waste in a landfill. It is also of importance when A may be accumulated, whether due to biological uptake or chemical processes (e.g., sorption onto soil, precipitation,etc.). Accumulated materials may be released at high concentrations if conditions change. With the four characteristics determined by the leaching test, the potential hazard of a waste can be evaluated, and the suitability of landfilling as a disposal technique assessed. The information from the test could also be used along with other information, to design waste processing or landfilling procedures so as to minimize the release of A from the waste.

A standard leaching test, as defined herein, will not give enough information to predict completely the four characteristics mentioned above; however, it will show the behavior of a waste under a prescribed set of conditions. These conditions can then be related to landfill conditions through modeling and careful selection of test conditions, and through correlation studies between test and actual landfill results. Correlation studies serve both to relate the non-modeled conditions to landfills and to verify the modeled conditions. With careful interpretation of results, an estimation of the behavior of a waste in a landfill can be made from standard test results.

Since the purpose of the test is to evaluate the leaching potential of a waste, it is reasonable to use an aggressive test to model a worst case situation. This is a conservative rationale. Further testing can then be done, if warranted, using conditions more typical of the long-term landfill situation.

A test which extracts only those components that would be leached in a landfill and extracts them in the same pattern that they would be extracted in a landfill is needed. A chemical solvent or series of solutions which would extract only leachable components would be ideal. Serial extraction procedures have been developed for soils and sediments; however, such procedures are based on a comprehensive understanding of the extraction process. This understanding requires that the chemical composition of the leached material and the chemical interactions between the material and the leaching solution be understood, and that the composition of the leached material be fairly consistent. Most of the wide array of landfilled wastes do not meet these criteria. The leaching test must then attempt to model landfill conditions, so the results are more readily related to full-scale landfill situations.

Batch and Column Tests

Two types of tests are commonly employed for determining the leaching potential of a landfilled waste—batch and column tests. In batch tests, a properly prepared sample of the waste to be tested is placed in a container along with leaching media. After a suitable period of time, and under conditions specified as being appropriate to the test, the elutriate or leachate is separated from the waste and analyzed to determine the material leached from the waste. Column tests, in which the waste is packed in a column and the leaching solution passed through, is a closer approximation of landfill conditions than a batch test, at least at first glance. The column test simulates both the waste—leachate contact (except around the column edge) and the rate of leachate migration found in landfills. The column test also is good for predicting the release pattern of A with time, since it models the continuous leaching and long time periods found in landfills. However, column tests have several disadvantages, such as the following:

1. problems arising from channeling and nonuniform packing,
2. potential unnatural clogging,
3. possibly unnatural biological effects,
4. edge effects,
5. long time requirements, and
6. difficulty in obtaining reproducible results even if done by experienced lab personnel.

All of these difficulties, but particularly the time requirements for an adequate column test (months to years), suggest that a batch test be chosen as the standard testing procedure. Both batch and column tests might be used in an intensive study.

Factors in a Batch Test

There are several factors affecting a parameter's concentration in the batch test elutriate which need to be considered in designing a leaching test. These factors are given in Table 1.

Even though specific values for each factor must be set or reported in order to obtain reproducible batch test results, the factors are interdependent, and values chosen for one may limit the range of reasonable values which may be selected for another. For example, a very high solid to liquid ratio may result in saturation concentrations for many elutions for species leached from a waste, unless the time per elution is shortened.

Of particular interest in the development of a leaching test are the test conditions. These are discussed individually below.

TABLE 1. FACTORS AFFECTING PARAMETER CONCENTRATIONS IN A BATCH TEST

1. Sample treatment prior to test
    - A. Sampling in industrial plant and shipment to test laboratory
    - B. Sample treatment in laboratory prior to test
        - 1. subsampling procedure
        - 2. sample preparation (solid-liquid separation, drying, grinding, etc.)
2. Test conditions
    - A. Leaching media composition
    - B. Solid to liquid ratio
    - C. Time per elution
    - D. Number of elutions
    - E. Temperature
    - F. Agitation technique
    - G. Surface area contact
3. Separation procedure and analysis after test

TABLE 2. CLASSIFICATION OF LANDFILLS AS RELATED TO LEACHATE COMPOSITION

Corresponding Landfill Type	Waste Landfilled	Leachate Composition Controlled by
Monolandfill	By itself, with relatively small amounts of other wastes, or with decomposed wastes	The waste itself
Municipal	With municipal wastes	Municipal refuse decomposition products
Industrial	With major amounts of other industrial wastes	Other industrial wastes

A. Leachate Composition

Three landfill situations represent the extremes in leaching media composition to which a waste might be subjected in a sanitary landfill, as shown in Table 2. Depending on the relative amounts of the waste being tested and municipal solid waste, and the extent of decomposition of the municipal waste, the appropriate leaching media may range in composition from leachate modeled on actively decomposing, municipal solid waste sanitary landfill leachate to something approaching distilled water. The latter would take on leaching characteristics from the waste itself, representing also the monolandfill situation in which the waste in question is disposed by itself. The third possibility is co-disposal with another industrial waste, which models leaching if the other waste controls the leaching media composition whether in a municipal or industrial waste landfill.

It is clear that the leaching media composition prior to waste contact is one of the key variables in a leaching test, and must be comparable to leaching media composition contacting the waste in the actual landfill for the test to have any practical significance.

For a monolandfill, distilled, deionized water or a synthetic rainwater can be used as a representative extractant or solvent. Alternatively, this landfill situation can be modeled by contacting the waste with a leachate produced by the waste itself. A sample can be contacted with leachate resulting from prior contact between another waste sample and distilled water, for example. This simulates what would occur if a drop of relatively uncontaminated rainwater passed through a thick layer of waste in a monolandfill.

For co-disposal with mixed municipal refuse, both distilled deionized water and a municipal landfill leachate can be used. The water simulates rainwater and provides mild leaching conditions. A municipal landfill leachate has widely differing characteristics depending on the refuse composition and state of decomposition, dimensions of the landfill, age, degree of chanelling of moisture, and both long-term and instantaneous climatic effects, etc. Further, even for a given sample of such leachate, the composition is very complex, precluding developing an exact recipe from which leachate of both reproducible and realistic composition can be produced. Rather than attempt to define a standard landfill, from which leachates representative of different landfill ages would conceivably be obtained, it is more promising to examine the leaching characteristics of leachate typical of actively decomposing municipal landfills, and model a synthetic leachate on the results. Such a synthetic municipal landfill leachate has been developed as part of this study, which simulates aggressive leaching conditions which might be obtained by co-disposal of the waste being tested with municipal refuse.

The leachate produced in a mixed municipal/industrial landfill is the most difficult to model, since the material entering the landfill will likely vary in composition. In this case, two approaches may be used in the selection of leaching media. If the wastes in the landfill are known, a synthetic leachate can be developed based on characteristics of those wastes. Alternatively, a series of leaching solutions can be used, each emphasizing a single leaching parameter—i.e., acid base, complexer, organic solvent, etc. Results obtained using different leaching solutions would indicate what types of wastes might be co-disposed with the waste in question. For example, a waste which released large amounts of an undesirable parameter under acidic leaching conditions should not be landfilled with acid or acid producing wastes.

The use of distilled water or other mild leaching solutions allows the waste to create its own leaching environment, whereas, a synthetic leachate or strong chemical solution essentially controls the leaching environment. For example, a waste containing small amounts of a leachable basic salt will raise the pH of a distilled water leachate, and only materials that are soluble in basic solutions will be found in the leachate. Conversely, use of a synthetic municipal leachate which is heavily buffered, or an acid leaching solution, will probably neutralize the basic salt while maintaining an acidic pH. In the first case the waste controls the pH of the solution, while in the second case the leaching media is the controlling factor.

B. Solid to Liquid Ratio

Solid to liquid ratios (or waste to eluent ratios) used in the test can have profound effects on test results. The concentration of a very soluble parameter will be directly dependent on the solid to liquid ratio (S/L). On the other hand, parameters for which concentrations are controlled by solubility will not show S/L ratio effects, but rather will have the same concentration at all S/L ratios, provided enough solid is present to saturate the system. S/L ratios can also affect concentration if adsorption or desorption processes are controlling the concentration.

In a given waste, several chemical constituents may be of interest. These may have different factors controlling their concentrations, and so may show different dependencies on the S/L ratio. Several currently available leaching or elutriation tests start with high S/L ratios and saturated conditions, then decrease the ratio until unsaturated conditions are reached. This procedure could be complicated if more than one parameter of interest reached saturation at different S/L ratios.

The S/L ratio encountered by a drop of leachate percolating through a landfill will be very high, by the very nature of percolation. If there were extensive channeling in the landfill, however, this would not be true.

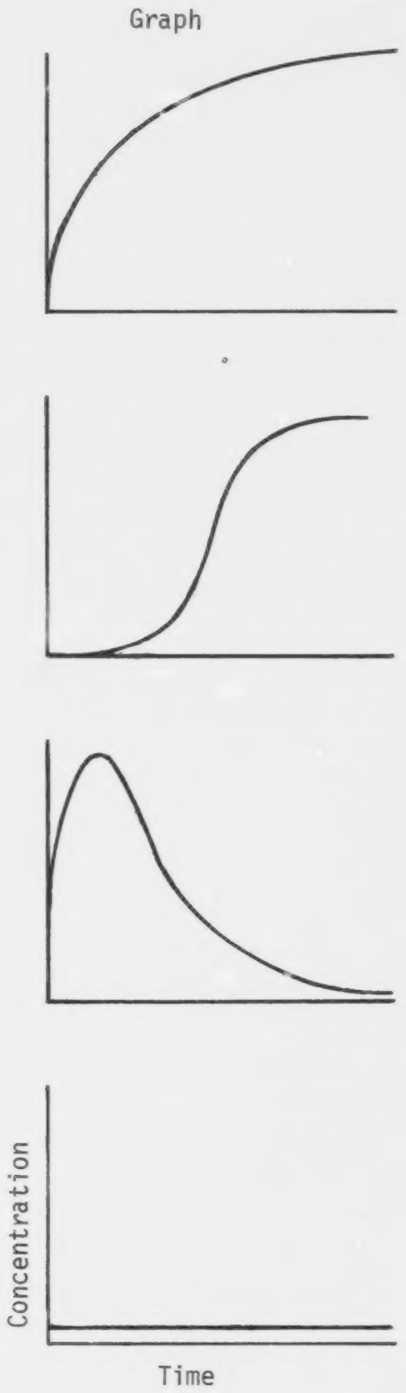
The choice of a solid/liquid ratio for use in the test is based on practical considerations. A very high S/L ratio, such as is used in the saturation test, is most likely to result in many components being saturated. This makes it difficult to estimate the total release of a component from the waste, since many elutions will be necessary to elute the leachable fraction of the component. Also, it is often difficult to obtain enough leachate for analysis with a high S/L ratio. On the other hand, a very low S/L ratio can produce very low concentations of the parameters of interest, leading to analytical problems.

An interesting approach to the selection of an S/L ratio is used in the State of Illinois E.P.A. test as reported in reference (1). The ratio to be used for Waste R is calculated from the equation R = 5.34 D, using the waste density (D) for the wet waste and a constant based on the average annual rainfall in Illinois. Thus, the S/L ratio is based on volume rather than weight, and can be readily interpreted in terms of landfill conditions and annual rainfall. If the results of a leaching test are to be directly related to landfill conditions, a correlation between the annual rainfall, the waste density, and the test S/L ratio will need to be made in order to determine how much leachate a unit area of waste will contact per unit time, and thus the time span to which the S/L ratio used in the test corresponds. The Illinois test is interesting in that the correlation is determined before rather than after the test is performed.

C. Time per Elution

Ideally, either each elution would allow the parameter of interest to come to equilibrium, or it would be designed to study the release kinetics of the parameter. In practice both situations are difficult to obtain. Different parameters may equilibrate at different rates. Lee and Plumb (2) found four release patterns in a leaching study using taconite tailings, as shown in Figure 1. The time span for their experiment was 500 days. The experiment used a very low S/L ratio (5 to 35 gm taconite per 10 liters distilled water), with periodic sampling. Not only did equilibration times for different parameters vary widely, but for some parameters a series of reactions occurred which produced concentration maxima with subsequent concentration decreases. The varieties of release patterns found make it apparent that no one sampling time could be chosen which is the best for each of the release patterns. Nor can a short leaching test be assumed to measure equilibrium concentrations of a parameter, or even to determine all the parameters that would be released from a waste. Of course, the four patterns found by Lee and Plumb are not the only release patterns possible; different wastes may have different and possibly unique release patterns.

The selection of an elution time is arbitrary. Some considerations to apply, however. The test should be long enough to allow rapidly equilibrating species to approach equilibrium and analytically determinable amount of most species to be released, yet short enough to minimize biological growth in the test chamber, secondary effects, and consistent saturation of species of interest. Biological growth can produce constantly changing conditions and could make test results very difficult to interpret. Consistent saturation of chemical species would require many elutions to be performed. Finally, the time chosen should be convenient to personnel, if possible.



Type	Example
Asymptotic Release	Specific Conductance, Alkalinity Ca, Mg and others
Exponential Release	Silica (a slow hydrolysis step needed before Si is solubilized)
Release followed by loss from solution.	Cu, Zn-loss due either to rising pH in solution, or absorption back onto solids
No Release	Several species

Figure 1. Types of release in long-term leaching test mentioned by Lee and Plumb, with examples from their leaching experiment using talconite tailings (2).

By removing samples at various times during an elution, one obtains information about the release kinetics of a rapidly equilibrating specie. However, unless the contact time between the waste and leachate in the landfill are known, this information may be difficult to apply to a landfill situation. It could be used to determine the equilibrium concentration of a parameter. The additional work involved makes this determination more appropriate in an intensive test rather than a standard leaching test.

D. Number of Elutions

The information obtained from more than one elution often justifies the extra work involved. Successive elutions can indicate the release pattern of a parameter over time, and often can give an idea of the factors affecting the release of the parameter. Successive elutions are particularly important when the release of one parameter, A, is inhibited by the release of another parameter, B. For example, imagine a leaching situation with a waste containing a soluble basic parameter (e.g., a carbonate) and an acid soluble-base insoluble component (e.g., a trace metal), being leached by an acidic leachate. The carbonates in the waste will neutralize the acid leachate until the carbonates have been leached from the waste. Incoming acidic leachate will then reestablish acidic conditions and bring the trace metal into solution. If only one elution was used, or if the test was ended before the acidic pH had been reestablished, the potential for trace metal leaching would be completely overlooked. More than one elution can also sometimes indicate the factors controlling the release of a soluble parameter—steady concentrations over several elutions may indicate solubility or desorption control, whereas, a rapidly falling concentration indicates washout.

The additional information obtained from repeated leachings needs to be balanced against the extra work involved. The experience of the authors is that the most useful information is obtained in the first several leachings. One reasonable approach is to set a set number of elutions, say three to five, with more elutions suggested if an indicator parameter, e.g., pH, has not returned to a baseline value.

In the discussion above, it has been assumed that the same waste sample has been eluted several times with fresh leachate. An alternative approach is to use the same leachate sample to elute several fresh samples of waste. This procedure provides information regarding the maximum concentrations that a parameter can reach in the leachate, rather than indicating release characteristics. By using both elution techniques—replacing either the leaching media or the waste in subsequent elutions—one can obtain considerably more information about the waste than by using either procedure alone.

E. Temperature

Temperature should have an effect on the leaching pattern of a waste due to its effects on solubility and reaction kinetics. Generally, however, leaching tests have been conducted at room temperature, and the effects of temperature on the leaching pattern of a waste within the range of average laboratory temperatures may not be great enough to justify specially controlled temperature conditions. Temperature should be measured and reported, however. Occasionally, temperature control can be very important if the waste itself is affected by temperature. For example, if a solid component of the waste melts at room temperature, constant temperature conditons are important.

F. Agitation Technique

An agitation technique which promotes mixing without causing waste particle or container abrasion is needed. Agitation is needed to avoid concentration gradients between the leachate in contact with the waste and that at a distance; however, overly vigorous agitation can cause particle abrasion (Boyle, et. al. (3)) and give unnaturally high results. One agitation technique, used in some tests, involves shaking for a short time followed by settling. The authors found that this procedure results in the development of significant concentration gradients between the settled waste and the leaching solution, and thus is not aggressive enough for a good leaching test. Other methods, such as reciprocal shaking, wrist action shaking or circular shaking are more suitable provided they produce well-mixed systems and are slow enough so as to not promote abrasion.

G. Surface Area Contact Between Waste and Leachate

For some wastes, the amount of surface area in contact with the leaching solution can be important in controlling parameter concentrations in the leachate. For example, viscous liquid or solid wastes which are water-impervious but which contain water soluble parameters, can show this behavior. Such species can be leached from the surface of the waste, where they are in contact with the water, but not from the interior of the waste since the waste is impervious to water. Diffusion through the waste is generally too slow for these species to reach the surface.

If this situation is known to occur with a particular waste, the surface area of the waste in the test should be measured and the release calculated per unit surface area as well as per gram. Interpretation of this data, however, may be difficult unless it is known how the waste is going to be landfilled and whether physical breakdown of the waste occurs with time. The surface area of a waste may be controlled initially during sample preparation before the test by grinding, cutting, etc., or by the agitation technique.

Summary of Existing Tests

Several batch leaching tests have been developed. A survey of some of the existing tests has been done by the Mitre Corporation (1). A summary of the surveyed tests (plus two additional tests) is given in Table 3. The table provides both the range and the frequency at which values occur within the range for each of the various test variables discussed in this section. For those factors for which the selection of a value is somewhat arbitrary, as in the S/L ratio or the elution time, the range of values reported might be given consideration in the specification of values to be used in a test, and an average value (median or mode) used. For other factors (especially the number of elutions, for example), average values have little meaning. The wide variety in all the specified factors indicates the need for a standardized test so that results on different wastes and by different laboratories would be comparable.

Concluding Statement

This section has discussed concepts leading to the general test procedure recommended as a result of this study. Also presented were discussions about each of the major test variables and how the selection of a particular value for each variable relates to the outcome of the procedure with a given waste. The remainder of this report will consider each aspect of the recommended leaching test in detail, starting with sample preparation and the solid-liquid separation. Next will be a discussion of each of the leaching procedure variables, including data representative of those used to study the effect of each variable on the leaching test results and eventually to set recommended operating levels for each variable. Finally, a section summarizing the recommended procedure and methods of data presentation and interpretation will conclude this report.

The amount of data obtained in this study is too much to incorporate in the body of the report without making it difficult to follow. Accordingly, examples of data are presented as appropriate, but the bulk of the data is presented in the Appendix. Typically, the Appendix presents results from different wastes or additional chemical species beyond those incorporated in the body of the report.

TABLE 3. SUMMARY OF EXISTING LEACHING TEST VARIABLES
(NUMBER OF TESTS SPECIFYING EACH OPERATING VARIABLE INDICATED)

Leachates							No.	
H_2O (dist, deion, dist-deion or unspecified)							17	
H_2O with pH adjustment or simple acid base							5	
Site specific							1	
Acetate buffer							1	
Synthetic municipal landfill leachate							1	
Synthetic natural rainwater							1	
Bacterial nutrient media							1	
Tests with more than one leachate							5	
Solid-liquid ratio	range 1:1-1:500	<1:4	1:4	1:5	1:10	>1:10	varied	calculated
		4	4	3	5	2	2	1
Time per elution	range 30 min- 10 days	<1 hr	1-24 hrs	24 hrs	48 hrs	72 hrs	>72 hrs	to "equil."
		1	3	7	3	2	3	2
No. of elutions	range 1-10	1	3	5	7	10		
		15	1	1	1	2		
Agitation	shaker, stirring & gas agitation used. Two tests use short agitation times with extended settling times.							
Surface area	unspecified							

SECTION 5

SAMPLE PREPARATION AND SOLID-LIQUID SEPARATION

Overview of Leaching Test Procedure and Wastes Tested

The organization of this report is such that the reader will not have a complete picture of the recommended leaching test until after a discussion of the many test variables. To avoid this, and to provide a basis for understanding the discussion and test procedures used in describing work related to each test element, it is necessary at this point to define in broad terms the batch leaching test procedure used and discuss the industrial wastes used. Details regarding the procedure, and the experimental background which determined the recommended operating conditions, will be presented throughout the remainder of this report.

Unless specified otherwise, leaching tests were run on whatever portion of a waste was not filterable through a 0.45 micron membrane filter. The waste was contacted with leaching media (normally distilled water) in a flask in the ratio of 1 part waste (wet weight) to 7 parts media, by weight. In later experiments, after it was found necessary to work with dry weight for consistency in working with a variety of wastes, a ratio of 1 part waste (dry weight) to 10 parts media was used. Usually 200 ml of media were required for each elution in order to obtain sufficient leachate for analytical purposes. The mixture was agitated by one of many methods evaluated during the study, all of which were designed to mix the waste and leaching media to promote the leaching process. After each 24 hour elution, the waste and leachate were separated with a 0.45 micron filter, the leachate analyzed, and the waste returned to the container and fresh media added for Procedure R. In Procedure C the waste was replaced with fresh waste and a portion of the leachate removed for analysis. Three elutions were performed, and standard analytical procedures were followed for leachate analysis as specified in Standard Methods (4). Most elemental analyses were done by atomic absorption. Because of the type of data read-out with this instrument, conentration values were obtained even though they were below the detection limits cited by the manufacturer. In such cases the measured values will be presented, as well as the cited detection limits, but they must be considered unreliable and approximate (e.g., Figure 25).

The wastes were obtained from several sources. They were selected to provide a range in chemical species and levels of solids contents. The wastes, and the symbols used in this report to identify them, were as follows:

1. shredded municipal refuse (residential and light commercial) obtained from the City of Madison shredding facility;

2. fly ash (unspecified further) from a power plant in Wisconsin;

3. paper mill sludge (N or unspecified further) from a paper mill in Wisconsin;

4. paint waste (AA) from an automobile assembly plant;
5. clarifier sludge (AA) from an automobile assembly plant;
6. fly ash (AA) from an automobile assembly plant;
7. fly ash (EPA) provided by EPA via Chem-Trol Pollution Services, Inc. of Model City, New York;
8. paper mill sludge (EPA) similarly provided by Chem-Trol;
9. a $CuO-Na_2SO_4$ slurry provided by Chem-Trol; and
10. wastewater treatment sludge (EPA) provided by Chem-Trol.

Most of the test development work utilized wastes 2 through 6 because these wastes were readily available in large amounts, and were expected to be reasonably constant in composition from one sampling date to the next. Fortunately, sufficient amounts of wastes were obtained the first time to avoid the need for resampling. Wastes 7 through 10 were supplied by EPA and were used only for major experiments to avoid the need for resampling. The wastes were stored in a cold room at 4°C. It should be noted that once developmental work had been accomplished and the recommended procedure established in the first phase of this research, many other wastes were tested to provide experience using a wider variety of wastes. The results of the second phase are reported separately.

Sample Preparation

Developing sampling and sample preparation techniques for obtaining representative and reproducible samples of industrial wastes was not a part of this project, except for the development of a solid/liquid separation procedure. Sample preparation is very important however. Standardized methods for sampling many industrial wastes are available, e.g., ASTM standards for various wastes, and it is suggested that these be used where possible.

In working with the various wastes used in the background study, a number of problems regarding sample preparation were experienced. These are discussed to illustrate some of the types of problems which may be encountered.

The problems encountered fall into the following main areas:

1. representative sampling,
2. particle size reduction,
3. leachate absorption by the solid,

4. homogenization, and

5. determination of dry weight.

These problems will be discussed separately.

A. Representative Sampling

If a waste consists of particles of different sizes, obtaining representative samples can be difficult. For example, the copper oxide-sodium sulfate slurry contained large crystals (softball size) in a thick, particle-laden slurry. In order to obtain representative samples, it was necessary to divide the whole bucket into the crystal and syrup fractions, weigh each fraction, grind the crystals with a mortar and pestle, then recombine the ground crystals and syrup in their original proportions. The problem of wastes containing large bulky solids in a slurry will probably not be uncommon.

Another example of a particle size related problem occurred with the fly ash samples. Although fly ash consists of relatively small particles, sieve analysis shows that the ash contains several different size fractions. The smallest particles are difficult to sample consistently when interspersed with the larger particles. One way to obtain consistent and representative samples is to separate the fly ash into different size fractions using a sieve, weigh each fraction, then recombine each sample according to the weight distribution of the original waste. A sand splitter may also be used for obtaining representative samples. The fly ash may need to be dried before sieve analysis.

The need for accurate particle size analysis and sampling is shown in Figures 2 and 3. Two fly ash particle sizes, less than 0.5 mm and between 0.5 and 1.00 mm, were leached with distilled water, using different types of agitation to be described later, and at different temperatures. It is apparent from the results presented in the figures that the pH, specific conductance and iron concentrations in the leachate are a function of the particle size of the fly ash. The elution rate is higher for the smaller particles, presumably due to the higher surface area of the smaller particles.

B. Particle Size Reduction

Bulky solids need to be reduced to smaller size, primarily to make handling easier and to promote leaching. A variety of different techniques can be used for the reduction. The crystallized solids in the copper oxide-sodium sulfate sludge were ground with a mortar and pestle. Some harder wastes, such as bottom ash, iron slag, etc. may need to be cut to proper size, perhaps a 1 cm diameter particle. However, if such wastes can be shown not to break down in a landfill, then the surface area of the waste in the test should be determined, and release calculated on the basis of surface area as well as weight. Evaluation of the leaching potential of such wastes should take into account the probable surface area of the waste in a landfill and the corresponding release.

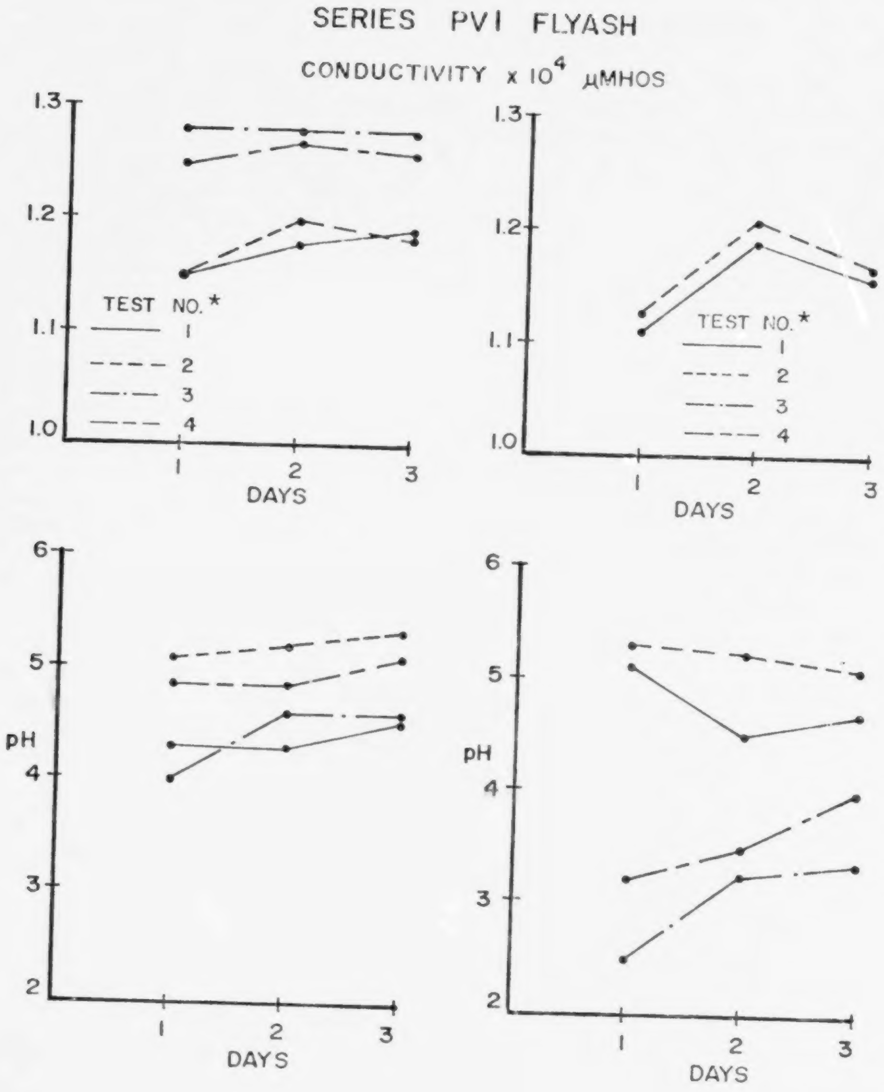


Figure 2. Differences in conductivity and pH in leachate from two fly ash particle size fractions.

*Note: see Figure 3 for key.

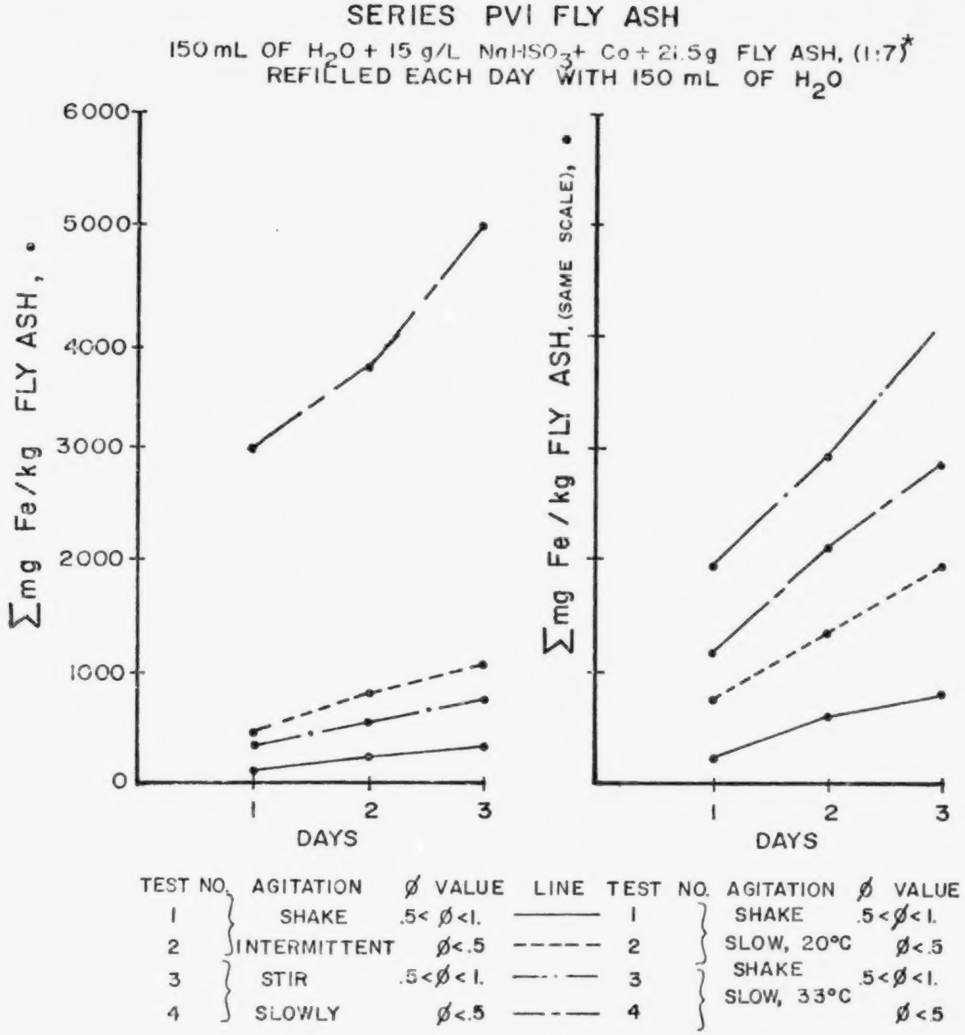


Figure 3. Differences in iron in leachate from two fly ash particle size fractions using a variety of agitation techniques.

*$NaHSO_3$ and Co (trace amounts) are to remove dissolved oxygen in the leach mixture.

Several processes for reducing leachability of waste have been developed, either through incorporating the waste in a solid matrix or by covering the waste with a stable impervious coating. These wastes may need special consideration with regard to sample preparation. If it can be shown that these wastes do not physically break down in a landfill or in the process of landfilling, the wastes should not be ground or cut up any more than necessary as the leaching characteristics of the ground waste in the test may be completely different from the characteristics of the containerized waste in the landfill.

Although not specifically tested as part of this study, it seems reasonable to cut, crush, or specially make wastes originally existing as large blocks to yield particles approximating the size equivalent to a 1 cm cube, for example. This particle size is small enough to work in the procedure, yet large enough to not increase drastically the surface area per unit weight waste exposed to leaching (see Figure 4). This is observed by plotting the surface area per volume contained for cubes of varying dimension ($a/v = 6s^2/s^3 = 6/s$). In interpreting the results, the use of a factor describing the change in surface area as a result of particle size reduction might be helpful. For example, reducing a unit weight of waste originally in 10 cm cubes to 1 cm cubes multiplies the exposed surface 10 times.

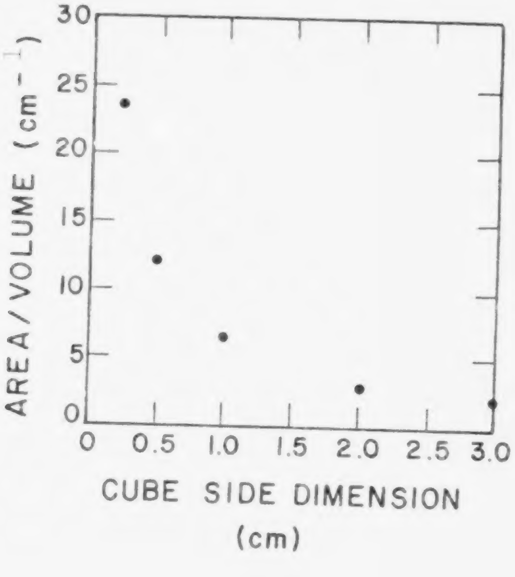


Figure 4. Effect of cubic particle size on surface area per unit volume particles.

C. Leaching Media Absorption by Wastes

Absorbant wastes that are significantly below field capacity may need to be brought to field capacity, either with distilled water or with the original liquid associated with the solid, if any. Otherwise, the leaching media may be absorbed by the waste and not available for leaching analysis. This was found to be a particular problem with paper mill sludge and the municipal waste where the latter had to be dried before grinding, then rewetted after grinding. Dry fly ash was also prewetted before leaching for this reason.

D. Homogenization

With a few wastes, a phase separation had occurred by the time the sample had reached the laboratory. One such waste used in the second phase of the study and not reported here was a coal tar sludge, in which the coal tar was floating on a water layer. The coal tar had the consistency of road tar, and stuck to anything it contacted. It would be next to impossible to homogenize the water and tar layers in this waste. Rather than homogenize the waste, the coal tar was used by itself in the leaching test, and the water layer could have been analyzed directly. The same situation occurred with a food waste and a grain-processing lipids waste, also tested in the second phase of the study. In both cases a solid organic material was floating on a water layer. Homogenization in such cases can be very difficult, and it is suggested that the organic portion be tested separately from the aqueous portion, in each case beginning with solid-liquid separation, etc. as appropriate.

E. Determination of Dry Weight

In order to accommodate wastes of different solids contents, to be able to compare the leaching potential of different wastes, and for leaching test procedural purposes, as will be discussed later, it is necessary to be able to define the dry weight of a waste. This turned out to be one of the more perplexing problems encountered. One would think that a routine procedure for determining the dry weight of a waste could be developed easily. With wastes containing non-volatile solids and water, the procedure is in fact routine. The waste is dried at a specified temperature either for a specified time or until constant weight is obtained (e.g., Standard Methods, 13 ed. (4)). Depending on the temperature, a portion of the water will be driven off (e.g., interstitial water, water of crystallization, etc.). However, for wastes containing volatile or semi-volatile organic compounds, the procedures given ambiguous results. The weight continues to drop even after long drying times as the partially volatile components are slowly driven off. Examples of this situation are given in Figures 5, 6, and 7 for three wastes (from phase 2). Even after 500 hours' drying at 105°C, the weight of these wastes continues to drop. It is not obvious what the dry weight of the waste is in these situations.

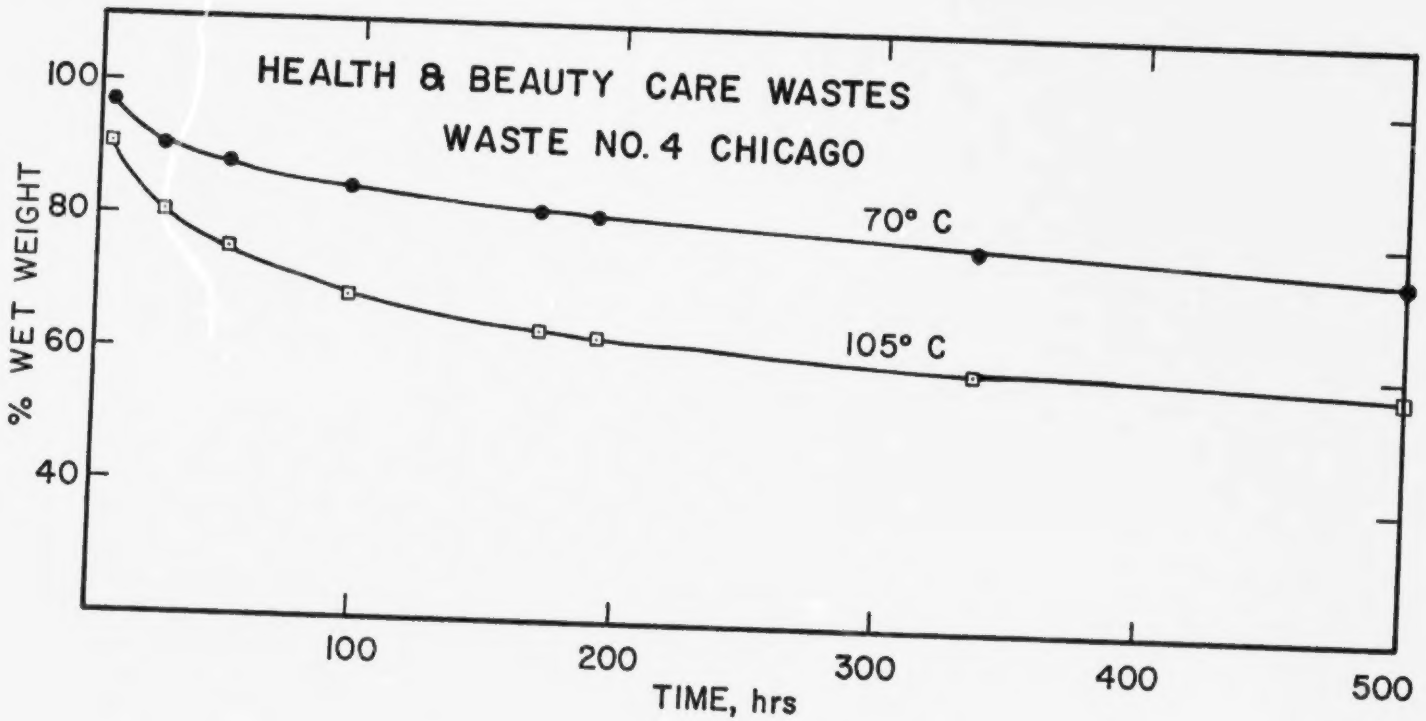


Figure 5. Long term drying characteristics of health and beauty care waste.

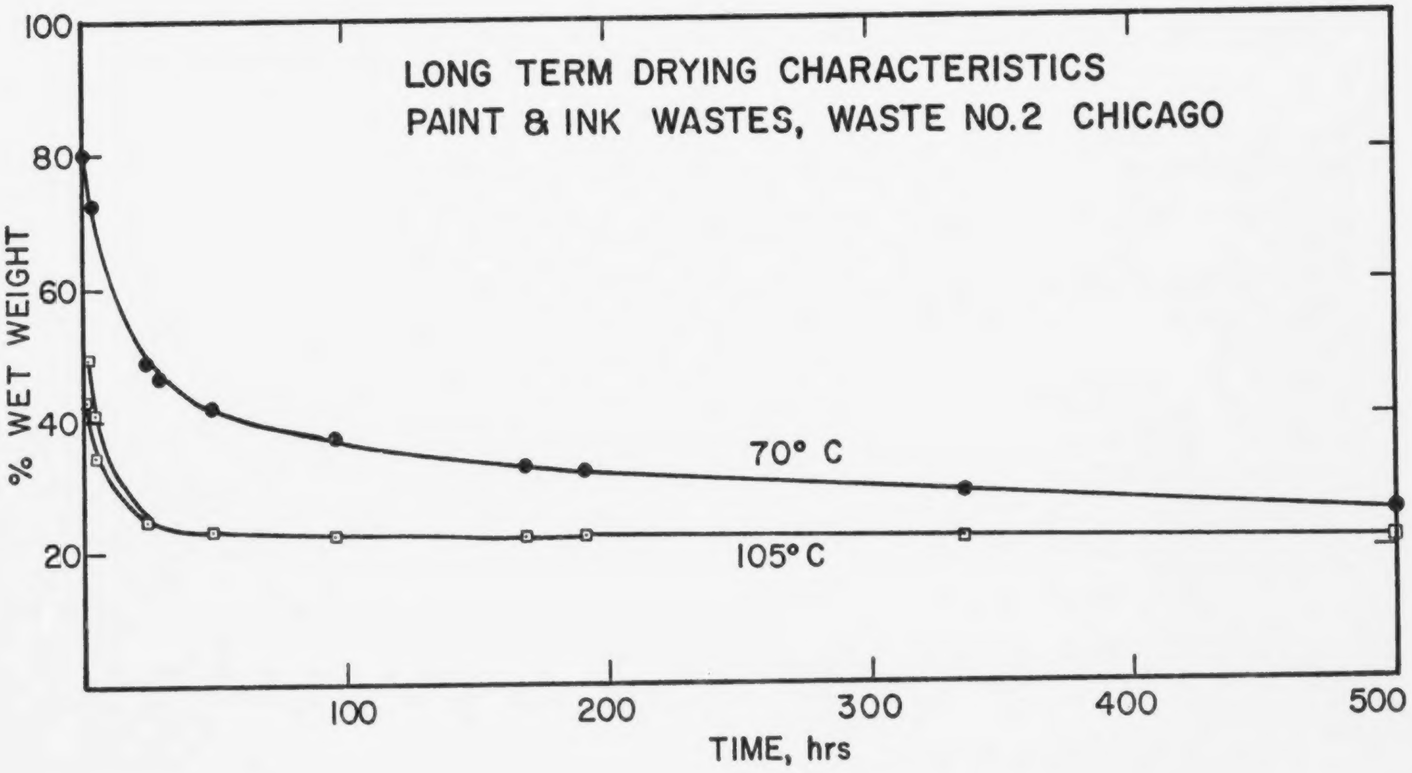


Figure 6. Long term drying characteristics of paint and ink waste.

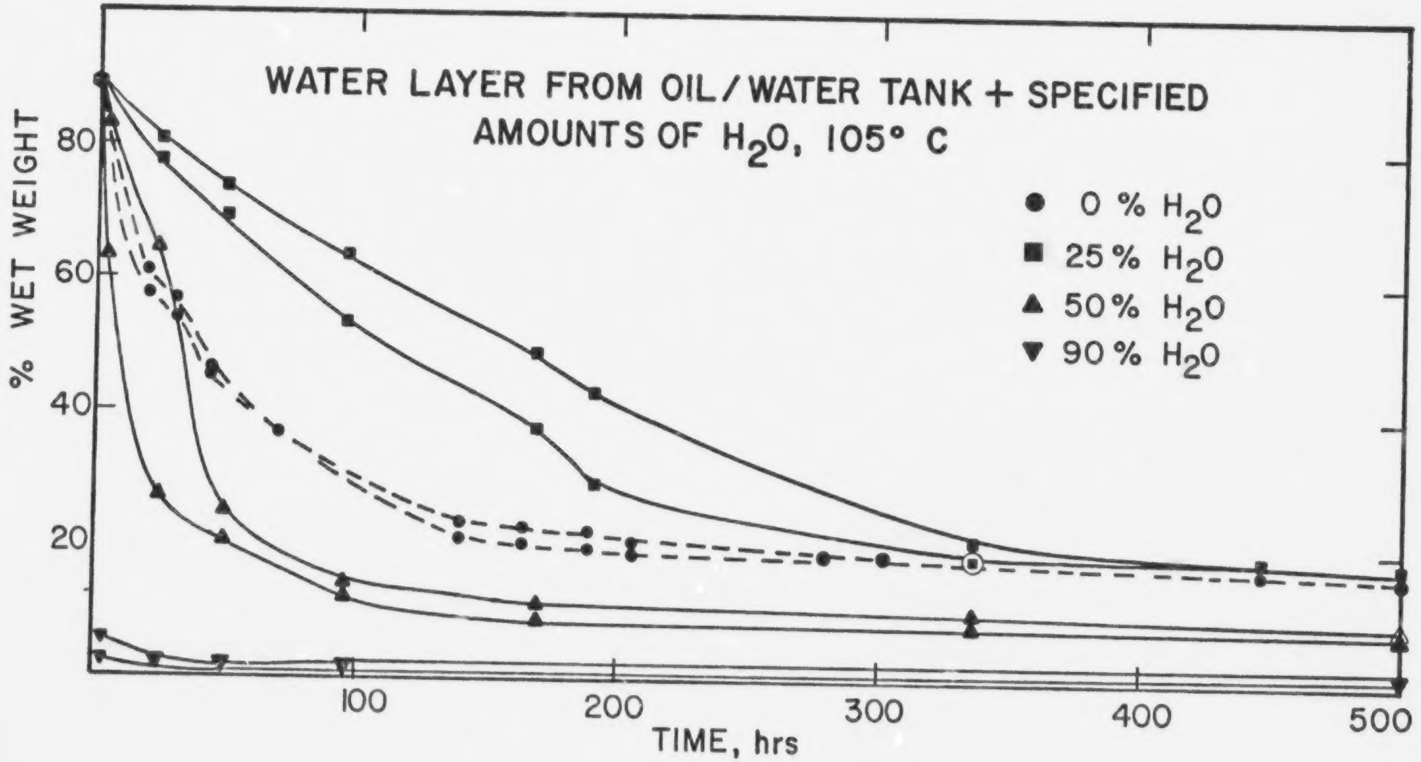


Figure 7. Long term drying characteristics of water layer from an oil-water tank with various amounts of water added.

Dry weight rather than wet weight is used to avoid problems of variable drying of the waste. For example, a 10 gm sample of a waste containing 50% water contains 5 gms of the waste itself. If the waste should dry so that it has a 10% water content, 10 gms of the waste now contain 9 gms dry weight. Since the release of materials from the waste is often dependent on the amount of waste present, these two samples might not behave the same in a leaching test. However, if a 10 gm dry weight of sample is used in the test, the same amount of waste is present no matter what the water content. The same problem can occur with volatile organic solvents as well as with water. Logically, to avoid problems due to drying, these solvents should be treated in the same manner as is water, i.e., removed from the sample before the determination of the weight. However, there is no clear distinction between volatile and nonvolatile organic compounds, as there is between common inorganic materials. It is interesting that the standard method suggested by ASTM for determining the moisture content in wood (D 1860 63 (reapproved 1976)) involves distillation rather than drying. Possibly, different drying procedures need to be prescribed for aqueous and organic samples.

Dry weights for the wastes tested were determined by drying at 105°C for 24 hours. Since the same procedure was used for each test, the results are comparable. For several wastes, particularly those mentioned in Figures 5 through 7, the dry weight is test condition specific, and is not an inherent property of the waste.

Solid-Liquid Separation

A preliminary step in the leaching test is the separation of the solid and liquid components of the waste. "Solid" and "liquid" in this context are defined by the separation. The rationale for the separation process is that the solid and liquid components of the waste will probably separate in a landfill. As illustrated in the following example, three separation processes might occur (Figure 8). After the waste is deposited in the landfill, the liquid components could flow downward due to gravity, be absorbed by surrounding materials, or move away from the waste by capillary action. In municipal refuse, the predominant identified material is often paper so that the absorption process is probably important. The solid material remaining after the liquid components have moved will be subjected to leaching by whatever leaching media is available in the landfill. Thus, it is more realistic to use only the solid portion of the waste in the leaching test, and to analyze the liquid portion separately, than to use the whole waste in the leaching test. The movement of the liquid portions of a waste from a landfill is not necessarily dependent on the leaching process.

Separations occurring in landfills will depend on the environment immediately adjacent to the waste and on the landfill conditions and design. Modeling such potentially varied conditions in the laboratory is very difficult. Therefore, it was considered more useful to develop a widely applicable and relatively easy solid-liquid separation scheme. Although the separation scheme is not unrealistic with regard to the separation that might occur in a landfill, it should not be considered an attempt to model that separation.

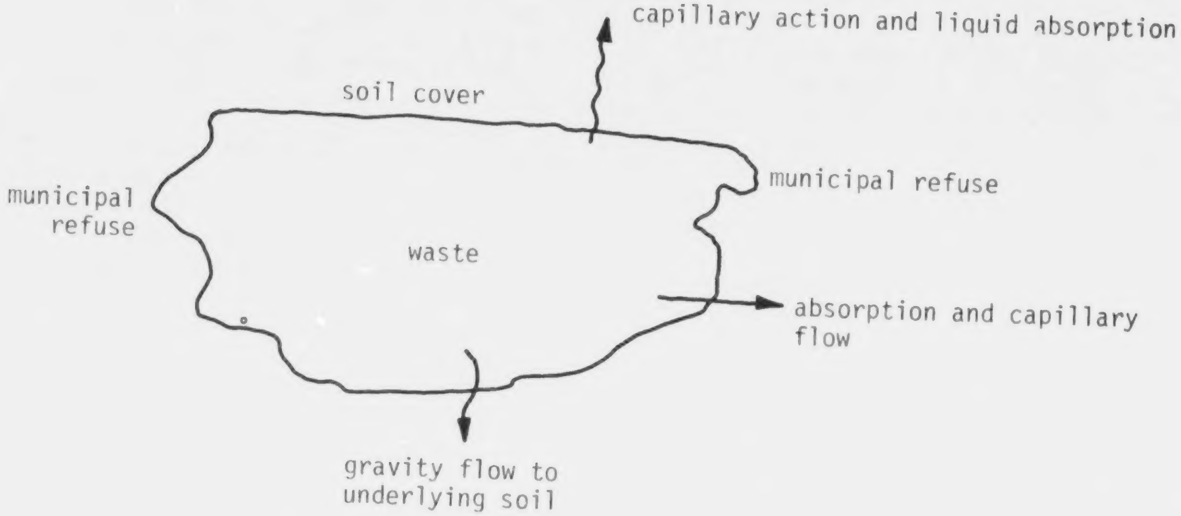

capillary action and liquid absorption
soil cover
municipal refuse
municipal refuse
waste
absorption and capillary flow
gravity flow to underlying soil

Figure 8. Movement of moisture from waste in a landfill.

TABLE 4. A List of Several Particle Separation Techniques

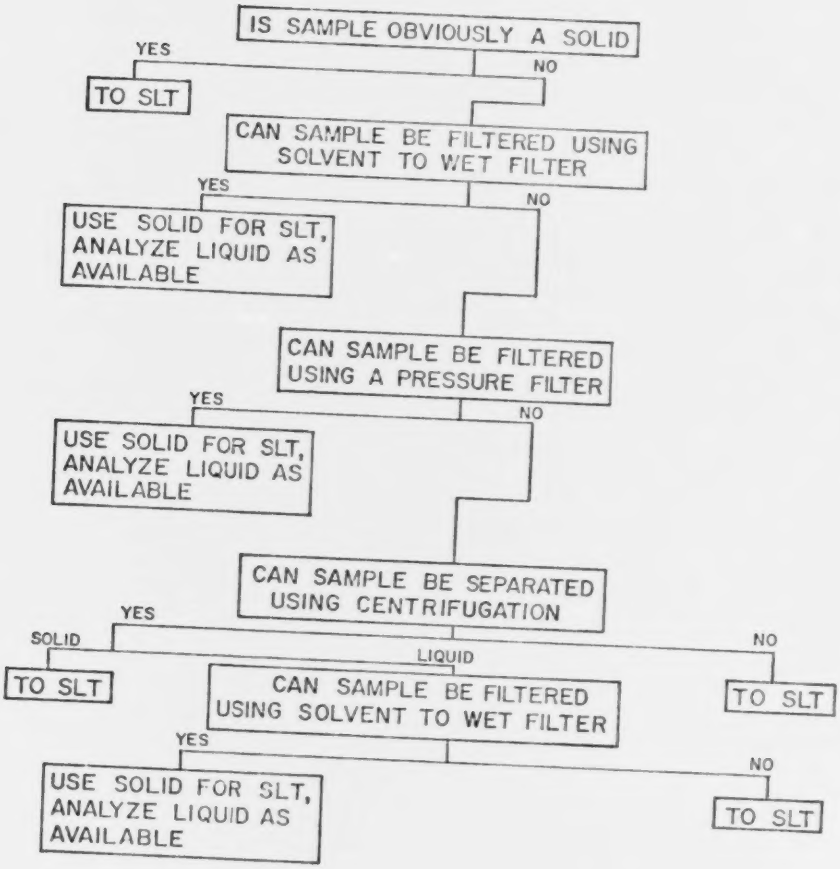
Filtration	Particle Electrophoresis
Sedimentation	Electrostatic Precipitation
Elutriation	Flotation
Centrifugation	Screening

Several particle separation techniques are given in Table 4. Of these, screening, filtration and centrifugation where chosen as being the most appropriate for the test scheme. Filtration was chosen as a final step in the scheme, since it is easily applied, readily available and standardized, inexpensive, and roughly approximates the separation processes in the landfill. Filtration operationally defines solids and liquids—anything that will pass through the filter is liquid, and all that does not is solid. It is important that the nature of waste components not be changed, but rather that they simply be separated. This precludes addition of coagulating or deemulsifying agents, for example.

The separation scheme, in Figure 9, can be described as follows: If the sample is not obviously a solid, an attempt is made to filter the waste through a 0.45 micron filter. If the liquid portion of the waste is not water soluble, the filter paper and support frit should be dry when the sample is placed in the filter allowing the liquid portion of the waste to wet both. If the sample will not filter, various other separation techniques are used to aid in the filtration. First, pressure filtration is employed using pressures up to 80 psi. This merely speeds up the filtration process and does not alter the nature of the separation. Anything that does not separate during filtration is centrifuged. If separation occurs during centrifugation, the liquid portion (centrifugate) is filtered through a 0.45 micron filter. Anything that will not filter after centrifugation or separate during centrifugation is considered a solid and is used in the leaching test. Occasionally, other techniques may be used if there are obvious reasons for their use. For example, sieving can be used to remove large particles that would result in clogged filters. In cases where the sample has previously separated through sedimentation or flotation, the solid and liquid portions of the separated waste can be treated separately. With all wastes, however, any liquid portion used for direct analysis must have been filtered through a 0.45 micron filter.

The selection of filter pore size is an important consideration. A small pore size will retain particles in the solid portion that might be considered liquid if a larger pore size were used. For example, hydrous ferric oxide (ferric hydroxide precipitate in water) precipitates in colloidal sized particles. A 0.45 micron pore sized filter will trap many of the colloidal sized particles in the solid portion, whereas a larger pore sized filter, e.g., 8.0 micron, will allow most of the colloidal sized particles to pass through the filter and so be considered liquid. Analysis for iron in the filtrates from the two pore sizes would give different values for the iron concentration in the "liquid" portion. An example of this situation is shown in Figure 10. A leachate containing hydrous ferric oxide precipitate was filtered through filters of different pore sizes, with the resulting "dissolved" iron concentrations presented in the figure. As can be seen, the dissolved iron concentration is much higher in the 8.0 micron and glass filter filtrate. Many materials may occur in or be associated with colloidal sized particles, so it is important to standardize the pore size used and to keep in mind the importance of the pore size on the designated liquid and solid fractions.

A filter pore size of 0.45 micron was selected for the final filtration step on the basis of its wide use in water and wastewater analysis, its availability in filters with carefully controlled pore sizes, and since it is not an unreasonable pore size for modeling landfill siutations. Particles larger than 0.45 micron occur in leachate, as evidenced by suspended solids measurements and the presence of bacteria, yet such materials are usually removed by passage through soils, as evidenced by the low suspended solids content of most groundwaters.


IS SAMPLE OBVIOUSLY A SOLID
YES
NO
TO SLT
CAN SAMPLE BE FILTERED USING SOLVENT TO WET FILTER
YES
NO
USE SOLID FOR SLT, ANALYZE LIQUID AS AVAILABLE
CAN SAMPLE BE FILTERED USING A PRESSURE FILTER
YES
NO
USE SOLID FOR SLT, ANALYZE LIQUID AS AVAILABLE
CAN SAMPLE BE SEPARATED USING CENTRIFUGATION
YES
NO
SOLID
LIQUID
TO SLT
CAN SAMPLE BE FILTERED USING SOLVENT TO WET FILTER
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NO
USE SOLID FOR SLT, ANALYZE LIQUID AS AVAILABLE
TO SLT

Figure 9. Solid-Liquid separation scheme.

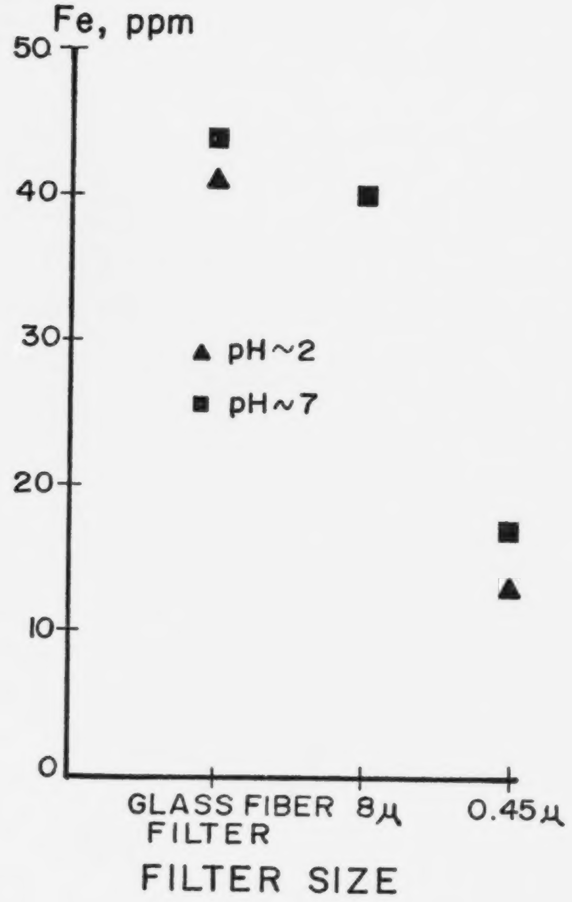

Figure 10. Comparison of the "dissolved" iron concentrations in municipal refuse leachate after filtration through various pore sizes.

The purpose of the centrifugation step is twofold: (1) to remove particles of diameter larger than approximately 0.45 micron from solution; and (2) to aid the removal of any emulsions that may be present in the wastes. Emulsions can be absorbed by dry solids in the same way a nonemulsified liquid can, yet will behave on a filter as a solid. The choice of centrifugation time and speed is made with these ojectives in mind. If the specific gravity of the particles are known, and if the suspending liquid is basically water, the centrifuge time and space needed to separate a 0.45 micron particle a distance equal to the length of the centrifuge tube can be calculated from Stoke's law and the Relative Centrifugal Force tables that accompany most centrifuges. If the particle density is unknown, a centrifuge speed of 2000 RPM for ten minutes for separating particles, and 10,000 RPM for ten minutes for separating emulsions is suggested.

The separation scheme has been applied to the wide variety of wastes used in the development of the leaching test. The wastes were successfully separated into solid and liquid components, with the solid components used subsequently in the leaching test. Use of unseparated wastes in the leaching test would have been much more difficult than use of solid components only, since many of the wastes were emulsion—solid mixtures which would be very difficult to separate and could absorb the leaching solution. If a solid-liquid separation is not performed prior to the actual leaching test, most if not all of the difficulties in performing such a separation are simply postponed to the point the leachate must be separated from the remaining solids for analysis after each elution. Thus, in addition to being more representative of landfill conditions, the separation also makes the leaching test easier to perform.

SECTION 6

INVESTIGATIONS AND DETERMINATION OF TEST CONDITIONS

Among the many test conditions and procedures which must be specified in a standardized leaching procedure, certain ones were deemed of sufficient importance to require specific investigations. They are listed below.

1. Leachate Composition
2. Solid-Liquid Ratio
3. Agitation Methods
4. Time per Elution
5. Number of Elutions
6. Temperature and Biological Aspects

These are discussed in order in the following sections for ease of presentation. It should be noted, however, that the test conditions and procedures are interdependent, and that laboratory work was not done to set one condition, then another, etc. Work was performed regarding the various conditions more or less simultaneously, with as many wastes as possible, in order to accumulate the experience and evidence necessary to set reasonable values or requirements for each.

Leaching Media Composition

A. Selection of Leaching Media

The original objective of this study was to quantify and model the leaching situation occurring in sanitary landfills. Because of changing leaching conditions as a result of decomposition process changes occurring as a landfill ages, and because of the likelihood of wastes other than municipal solid wastes being present in a landfill and affecting leaching conditions for the waste being tested, it was felt necessary to consider some of the extreme situations which could occur. If the subject waste is landfilled with only municipal solid waste, it could be subject to leaching media ranging in composition from leachate typical of young (actively decomposing) to aged (stabilized) municipal waste sanitary landfills. Further, depending on the relative amounts of the subject waste and municipal solid waste, the leaching media will take on characteristics ranging from municipal waste landfill leachate to leachate arising from the industrial waste itself, as in a monolandfill. Finally, since many wastes are particularly susceptible to certain leaching conditions (e.g., inorganic components are generally more readily leached under acidic pH conditions), and since wastes other than normal municipal (residential and light commercial) solid wastes may be in a municipal landfill as well, it was felt necessary to incorporate sufficient flexibility in the leaching test procedure to examine the effect on leaching of the subject waste of peculiar leaching conditions, as would be possible if other industrial wastes would control leaching media composition.

The extremes of the landfill situations discussed above are represented by monolandfill, sanitary landfill, and industrial landfill situations, as shown in Table 5. The monolandfill situation describes landfilling of the waste in question by itself, or in a landfill where it controls the leaching media composition either by being present in large amounts, or the municipal refuse being basically decomposed. An appropriate leaching media in this situation can be obtained by contact of distilled water and the subject waste, by which the distilled water takes on characteristics derived from the waste itself. Leaching media modeled on actively decomposing municipal solid waste landfill leachate relates to leaching of the industrial waste in an actively decomposing sanitary landfill, where the municipal solid waste controls leachate composition. Co-disposal with another industrial waste represents the extreme situation if the leachate characteristics are controlled by another waste, whether it be in a so-called municipal waste sanitary landfill, or in co-disposal in an industrial waste landfill. The leaching solution for the industrial landfill could either be based on the composition of the landfilled wastes and hence the leachate, if known, or consist of a number of leaching solutions. The different leaching solutions would emphasize a single leaching parameter—acid, base, complexing, etc. A reasonable leaching media could also be obtained by contacting the other wastes with distilled water, separating the spent waste, and using the resulting leachate as the leaching media in contact with the waste under study. If biodegradation is likely, a synthetic or laboratory-derived leachate may be desired, similar to that which will be discussed for co-disposal with municipal refuse. The results from the different leaching

TABLE 5. LEACHING SOLUTION FOR VARIOUS TYPES OF LANDFILLS

Representative Landfill Situation	Leaching Solution
Monolandfill	Distilled, deionized water or leachate produced from the waste itself.
Sanitary landfill	Municipal refuse sanitary landfill leachate.
Industrial landfill	Either a leachate based on the characteristics of the wastes in the landfill or a series of leaching solutions as described in the text.

solutions would indicate with what other types of wastes the waste in question could or could not be landfilled. Thus, a waste which releases large amounts of an undesirable parameter under acid leaching conditions should not be landfilled with an acid or acid-producing waste.

The emphasis of the remainder of this section is the development of a municipal refuse sanitary landfill leachate, to model co-disposal of the waste in question with municipal refuse.

B. Development of a Synthetic Municipal Landfill Leachate

To model the leaching occurring in municipal landfills, a synthetic municipal landfill leachate (hereafter called synthetic leachate) is needed. The characteristics of municipal landfill leachate are such that it cannot be modeled adequately by distilled water or by a simple solution (e.g., 0.1 N acid). Rather, the aggressive characteristics of the real municipal leachate need to be analyzed, and a synthetic leachate modeling these characteristics developed. While it might be possible to produce a real leachate in the laboratory, using municipal refuse and water, the difficulty of producing reproducible solutions, the instability of real leachate, and the high and often unstable background values for many parameters of interest, favor the use of a more stable and controlled synthetic leachate. The designation of a "standard" landfill, from which a "standard" leachate could be obtained, is likewise unrealistic.

Unfortunately for the modeler, municipal landfill leachates are far from being homogeneous solutions. The pH can range from acidic (<4) to basic (>8), while the measured ranges for many other parameters can span six orders of magnitude or more. Obviously, one synthetic leachate cannot model all municipal leachates. Since it is the aggressiveness of the leachate that is of interest, it is reasonable to model leachates of maximum aggressiveness.

The model development was as follows:

1. aggressive parameters in leachate were identified,

2. landfill degradation processes affecting these parameters were studied,

3. maximum measured concentrations of the parameters were obtained from the literature and model compounds were chosen,

4. the compounds were combined into a synthetic leachate at concentrations based on the common maximum concentrations in real leachate.

The maximum likely concentrations rather than the mean or weighted average concentrations were used to give a reasonable worst case analysis. In order to model the maximum leachability of industrial wastes, the chemical parameters in leachate representing the most aggressive leaching media (e.g., low pH, low redox potential, high complexation ability, etc) the waste would probably encounter were chosen. It is necessary to keep this in mind when interpreting the results.

(1) Aggressive parameters considered

Of the parameters judged to be of importance in bringing materials into solution from solids, the following were chosen for consideration:

pH
complexing capacity
redox status
organic solvents
ionic strength

During the aging cycle of a landfill, these parameters will vary in strength due to changes in the concentrations of materials producing them. In order to understand and evaluate the variations found in the parameters being considered for the synthetic leachate, some understanding of the processes occurring in landfills is necessary.

(2) Theoretical degradation of a landfill

Imagine a hypothetical landfill with no external influences except for a constant water input. As the landfill ages, a succession of stages will occur.

Little or no leachate is produced until the landfill reaches field capacity (becomes saturated with water). The composition of any leachate produced prior to saturation, due to compaction and squeezing, will depend on the composition of the water initially landfilled, and may vary greatly.

Three major bacterial processes primarily responsible for degrading refuse are shown in Figure 11. Initially, aerobic decomposition predominates. This phase will generally be very short, given the limited amount of oxygen in the landfill and the high BOD of the refuse. During this phase, a large amount of heat is produced, raising the landfill temperature well above ambient temperature.* Leachate produced during this phase would be expected to dissolve very soluble salts (e.g., NaCl) landfilled with the refuse.

As oxygen is depleted, decomposition caused by facultative anaerobic bacteria will predominate. During this first stage of anaerobic degradation, large amounts of volatile fatty acids (e.g., acetic acid) and CO_2 are produced. These acids reduce the pH to low as 4.5 to 5. The low pH helps to solubilize inorganic materials which, along with the high volatile acid concentrations, produce a high ionic strength (specific conductance). The high volatile acid concentrations also contribute to the high CODs often found during this phase. The oxidation-reduction potential (redox is reduced to below 0 mv (with respect to a Standard Calomel Electrode).

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*Assuming an initial temprature high enough to start the degradation processes. One landfill, started during the winter, was found to be still partially frozen over a year later.

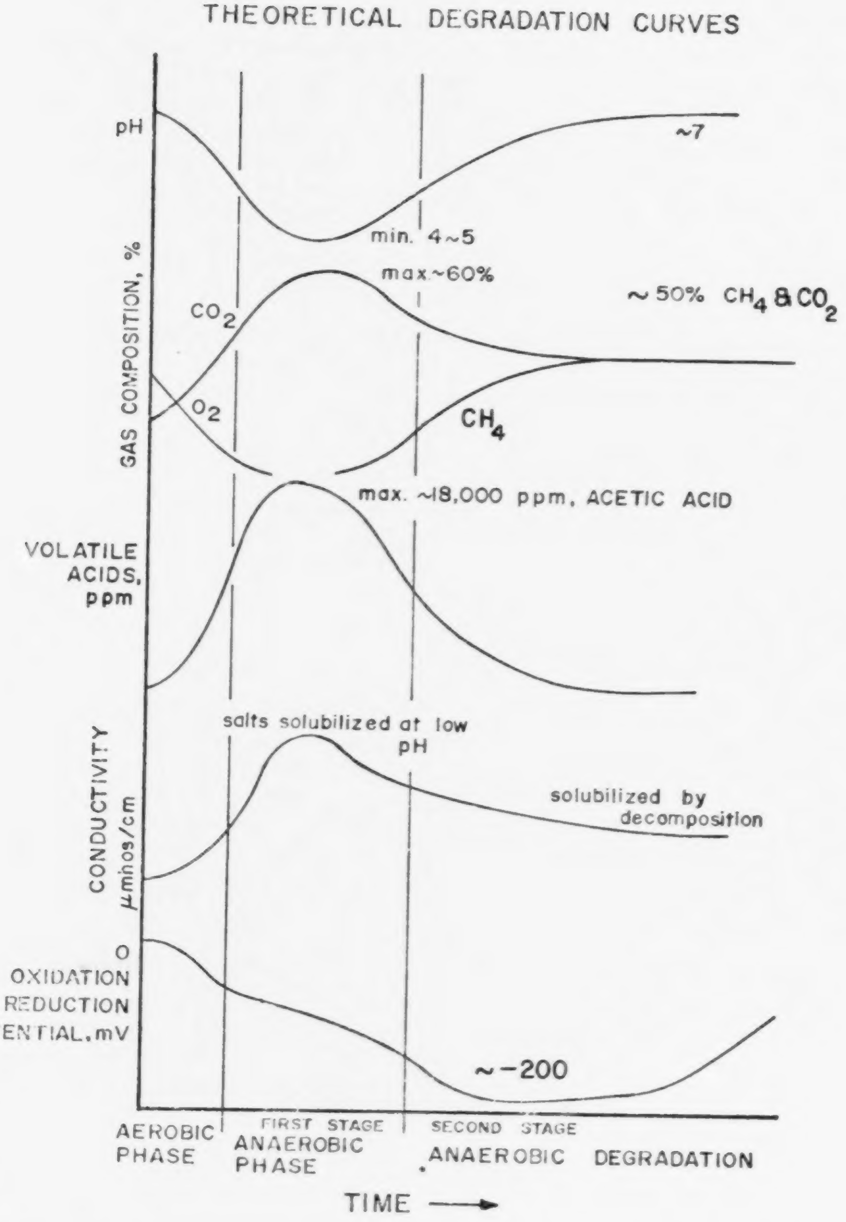

Figure 11. Theoretical degradation curves of a theoretical landfill.

The second stage of anaerobic decomposition occurs when methane producing bacteria complement the facultative anaerobes. Methane bacteria are strict anaerobes and require neutral pH levels. Volatile acids produced by facultative anaerobes and other organic matter are converted to methane and CO_2. Thus, the volatile acid concentration is reduced, and the gas composition becomes a mixture of CO_2 and CH_4. With the neutral pH necessary for the bacteria to live, fewer inorganic materials will be solubilized, and specific conductance will fall. The redox potential should be lower than the potential during the first stage of anaerobic processes, reflecting the low potential needed for methane production and the higher pH. Eventually, bacterial action may decrease as the substrate is depleted and higher redox potentials may be reestablished by oxygenated water.

(3) Actual degradation of a landfill

Environmental conditions may alter the theoretical degradation pattern considerably. The amount of water input has a very important effect on the rate of degradation. Obviously, the composition of the refuse landfilled also has important effects as do landfilling practices and seasonal variations in temperature. To complicate matters further, different microenvironments in the landfill may undergo different stages of decomposition at the same time. For example, Emcon Associates (5) found high volatile acid production, low pH, and methane production occurring simultaneously. Since the low pH is toxic to the methane producing bacteria, it is apparent that different areas of the landfill had different and mutually exclusive conditions, with the leachate reflecting both.

The data used in evaluating the parameters of interest came primarily from the relatively few studies that have done detailed analysis on leachate from a single landfill over a long time period (6-9), or from work by Chian et al (10) relating leachate composition from different landfills to landfill age. Analysis of a single leachate sample from a landfill was generally found not to be very useful, since the concentration of a given parameter could not be related to the aging process in the landfill.

Chian et al (10) analyzed several classes of organic compounds and related variations in their concentrations to landfill age. Figure 12, based on their work, shows the variations of these classes as a percentage of the total organic carbon with landfill age. The age axis should be regarded as approximate, since landfill degradation rates vary with environmental conditions. Their results will be discussed in the sections below dealing with the aggressive parameters.

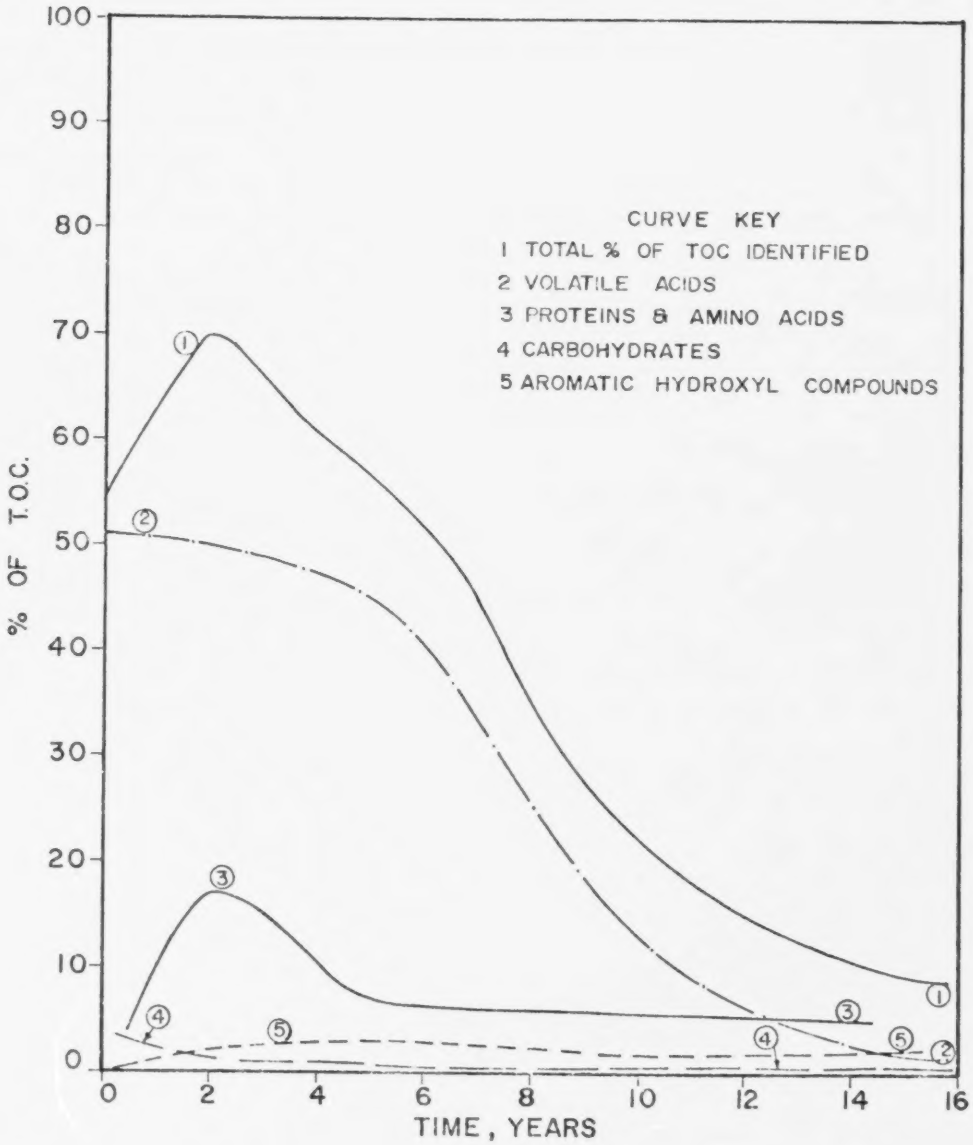

Figure 12. The trends in the identified fractions of leachate TOC vs. the age of the landfill. (From Reference (10).)

(4) Maximum measured concentrations of the parameters and model compound selection

—pH

There are two factors of importance in modeling pH and redox; the measured value and the buffering capacity that maintains that value. The buffering capacity indicates how resistant the measured value will be to change. The minimum pH found in leachate comes during the period of volatile acid production in first stage anaerobic decomposition. Chian et al (10) show that the pH and volatile acid trends in real landfills follow the theoretical trends fairly closely. The pH commonly reaches four or five and is heavily buffered by volatile acids Table 6 gives the pH ranges reported by various authors from their literature reviews, while Table 7 gives the minimum pH values reported in various studies. A pH of 4.5 was selected for the synthetic leachate. As can be seen from Tables 6 and 7, a pH of 4.5 is not uncommon in leachates. Furthermore, both CO_2 and volatile acid solutions achieve maximum buffering capacity near this pH. An "average" landfill probably does not maintain this low a pH for an extended period of time, but, rather, maintains a pH of between 5 and 5.5. The emphasis here is leachate aggressiveness, which warrants the use of the low pH value.

TABLE 6. pH RANGES REPORTED BY VARIOUS AUTHORS FROM LANDFILL OR LITERATURE SURVEYS

Source	Range
Chian et al (10)	3.7 - 8.5
Steiner et al (11)	4.0 - 8.5
Clark et al (12)	1.5* - 9.5
Encom Associates (6)	3.0 - 8.5
Pohland (7)	4.9 - 8.4

*Site received acidic industrial wastes

Relatively few investigators have studied volatile acid concentrations in leachate. From the data available, given in Table 8, a value of 300 mM (or 18,000 mg/l as acetic acid) was selected as a reasonable approximation for the highest concentrations commonly found in leachates. Acetic acid is used as a representative volatile acid, since it is generally the acid in highest concentration in leachate during this stage of landfill decomposition. Since the concern is with pH and buffering capacity, use of a single acid to model the different volatile acids poses no problem. The synthetic leachate contains 0.15 M acetic acid and 0.15 M sodium acetate. The sodium is additionaly used in modeling ionic strength.

TABLE 7. MINIMUM pH VALUES FOUND IN LEACHATE STUDIES

Source		pH
Emcon Associates (5,6)	A	4.6
	B	4.2
	C	4.3
Qasim et al (9)		5.3
Fungaroli (8)		5.0
Pohland, Fill 1 (7)		5.1
Hughes et al (13)		6.5
U. IL*		5.6
Boon Co. *		5.3
Madison MUNC*		6.0
Madison UMC*		5.6

*From Chian et al (10). Single leachate sample.

TABLE 8. VOLATILE ACID CONCENTRATIONS FOUND IN LEACHATE

Landfill or Author	Concentration, mM/l
Emcon Associates (5,6)	
Cell A*	456
Cell B*	307
Cell C*	192
Pohland* (7)	155
Madison, MUNC†	300
Madison, UMC†	26
Boon County†	73
U. IL**	200
Burrows et al (14)	850‡

*Maximum.

†From Chian et al (10).

‡Value seems very high, especially since no acetic acid was found found in the leachate.

—complexation

Many different ligands, both organic and inorganic, can complex metals and leach them from industrial wastes. Organic compounds containing nitrogen, oxygen, or sulfur in the proper configuration can be very strong complexers. Data is available on the aging patterns of aromatic hydroxyl groups in humic and fulvic acids and of N in proteins and amino acids (Figure 12). Other complexing ligands may be of importance in leachate, but the data available is not sufficient for modeling.

Humic and fulvic acids are general terms given to the heterogeneous aromatic acids found in natural waters. They are often of high molecular weight and may contain N or S groups. Since these materials are of heterogeneous composition, they are somewhat difficult to model with a single compound. Chian et al (10) estimate that approximately 3% of the TOC of leachates is in aromatic hydroxyl compounds (see Figure 12). Based on the leachate data available, given in Table 9, a value for the

TABLE 9. CALCULATED CARBON CONCENTRATIONS OF AROMATIC HYDROXYL COMPOUNDS FOUND IN LEACHATE

As 3% TOC, from Chian et al (10)	
Sample	(mMC)
MUNC	69.2
UMC	14.8
Boon County	34.6
U. IL	42.7
Calculated from Tannin and Lignin Data, Qasim et al (9)	
Cyl. A	17.4
Cyl. B	33.8
Cyl. C	50.0
VALUE USED	50.0 mM C

fulvic acid associated carbon of 0.05 M seems a reasonable approximation of the fulvic and humic acid hydroxyl group concentration. The actual concentration of the compound used to model these acids depends on the carbon content of that compound.

Several aromatic hydroxyl compounds were tested as suitable complexing agents. Tannic acid ($C_{76}H_{52}O_{46}$) and gallic acid (3,4,5-trihydroxybenzoic acid) were found to precipitate with time. Pyrogallol (1,2,3-trihydroxybenzene) was tried and was found to be an excellent complexer, and a good reducing agent, as well. It is stable in solution if kept from air contact. Other compounds were tried, including salicylic acid, α-resorcylic acid (3,5-dihydroxybenzoic acid), benzoquinone, quinone, and phthalic acid, but none were found to have the solubility and stability in solution and the reducing capability of pyrogallol. Pyrogallol is included in the synthetic leachate at 8mM.

The other complexing group being modeled is the amine group on proteins and amino acids. Chian et al (10) found that a maximum percentage of the TOC as amino acids and proteins occurred early in the life of the landfill (Figure 12). They ascribed this peak to single amino acids. Glycine ($H_2NCH_2CO_2H$), the simplest of the amino acids, was selected to model organic N in leachate. The concentration used, 50 mM, is based on organic N concentrations in leachate found by various researchers, which are given in Table 10. This assumes that all the organic N is in amino acid form.

TABLE 10. ORGANIC NITROGREN CONCENTRATIONS FOUND IN LEACHATE

Landfill	Concentration, mM
Pohland (7), Fill 1	12.6
Fungaroli (8)	15.0*
Emcon Assoc. (5,6)	
A	35.9
B	55.3
C	57.1
Madison, MUNC†	73.4
Madison, UMC†	5.6
Boon Co.†	2.2
U. IL†	38.9
VALUE USED	50.

*Ignoring an initial peak.

†From Chian et al (10).

—redox potential

Reactions controlling redox potential in leachate are not as well understood as those controlling pH, and are correspondingly more difficult to model. Bacteria in the landfill and in the leachate itself play a major role in controlling redox potential. Not only do bacterial processes in the landfill reduce the redox potential of the leachate, but bacteria in the leachate itself appear to maintain a low redox potential in the leachate even after it is separated from the landfill. It was found that the redox potential in unfiltered leachate would maintain a steadier value when exposed to air than would filtered leachate, as shown in Figure 13. Further, adding phenol to the unfiltered leachate to inhibit or kill the bacteria caused the redox potential to rise rapidly, paralleling the rise in the unfiltered leachate. Stirring also had an effect. If the unfiltered leachate were allowed to settle, the redox potential in the supernatant rose. Upon stirring, the redox potential fell to around its previous value. One plausible explanation for this behavior is that the facultative anaerobes in the unfiltered leachate remove any oxygen that enters the sample and, thus, maintain anaerobic conditions. Killing microorganisms with phenol or allowing them to settle out allows oxygen to enter and raise the redox potential. In both filtered and unfiltered samples, a brown flocculant precipitate was present, presumably an iron salt, indicating that the potential measured in both samples may not have been the potential found in the landfill.

Similarly to modeling pH, it is important not only to model the redox potential itself, but also to model the redox buffering capacity. While it is possible chemically to model redox potentials found in leachates, it is quite difficult to model the buffering capacity produced by the bacteria. The ability of bacteria to maintain a redox potential when in contact with an industrial waste may be quite different from the ability of a chemical redox couple to sustain the same redox potential. Chian et al (10) predicted a redox potential between 0 and -200 mv for leachate from a young landfill.

A pyrogallol - Fe^{2+} complex was selected to control redox potential in the synthetic leachate. Pyrogallol is used also as a complexing agent and, as such, its concentration is set at 8 mM. Ferrous iron was chosen because it is a good redox control agent, especially when complexed with the pyrogallol, is natural to leachates in high concentrations, and can be used to model divalent cations. Other metals and complexing agents were tried as redox control agents, including Sn^{2+}, Mn^{2+}, ascorbic acid, and the complexing agents mentioned previously. None of these gave the desired redox potentials in synthetic leachate. A Fe^{2+} concentration of 24 mM was chosen to give a redox potential of -30 mV (vs SCE) at a pH of 4.5. Redox potential varies with pH, as shown in Figure 14, and the -30 mv quickly drops to lower values as the pH rises.

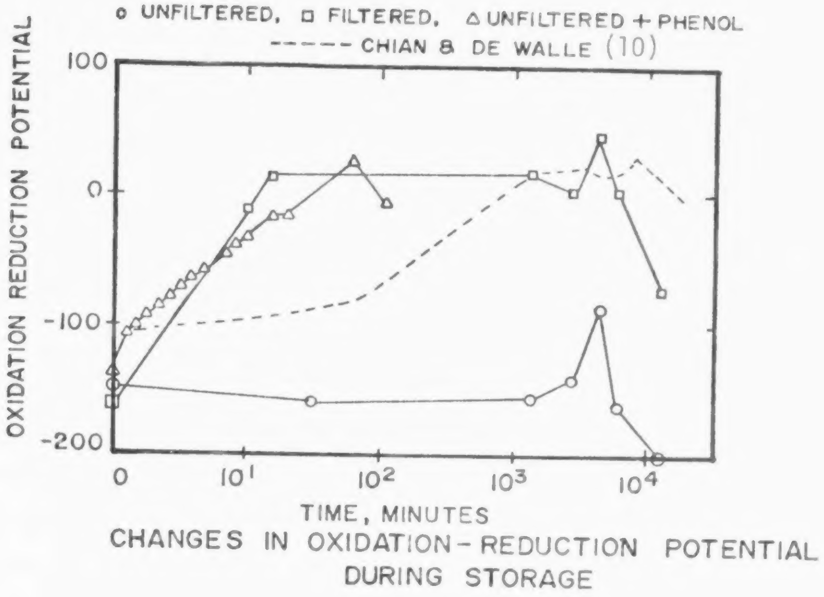

Figure 13. Changes in the redox potential of leachate during storage and after filtation.

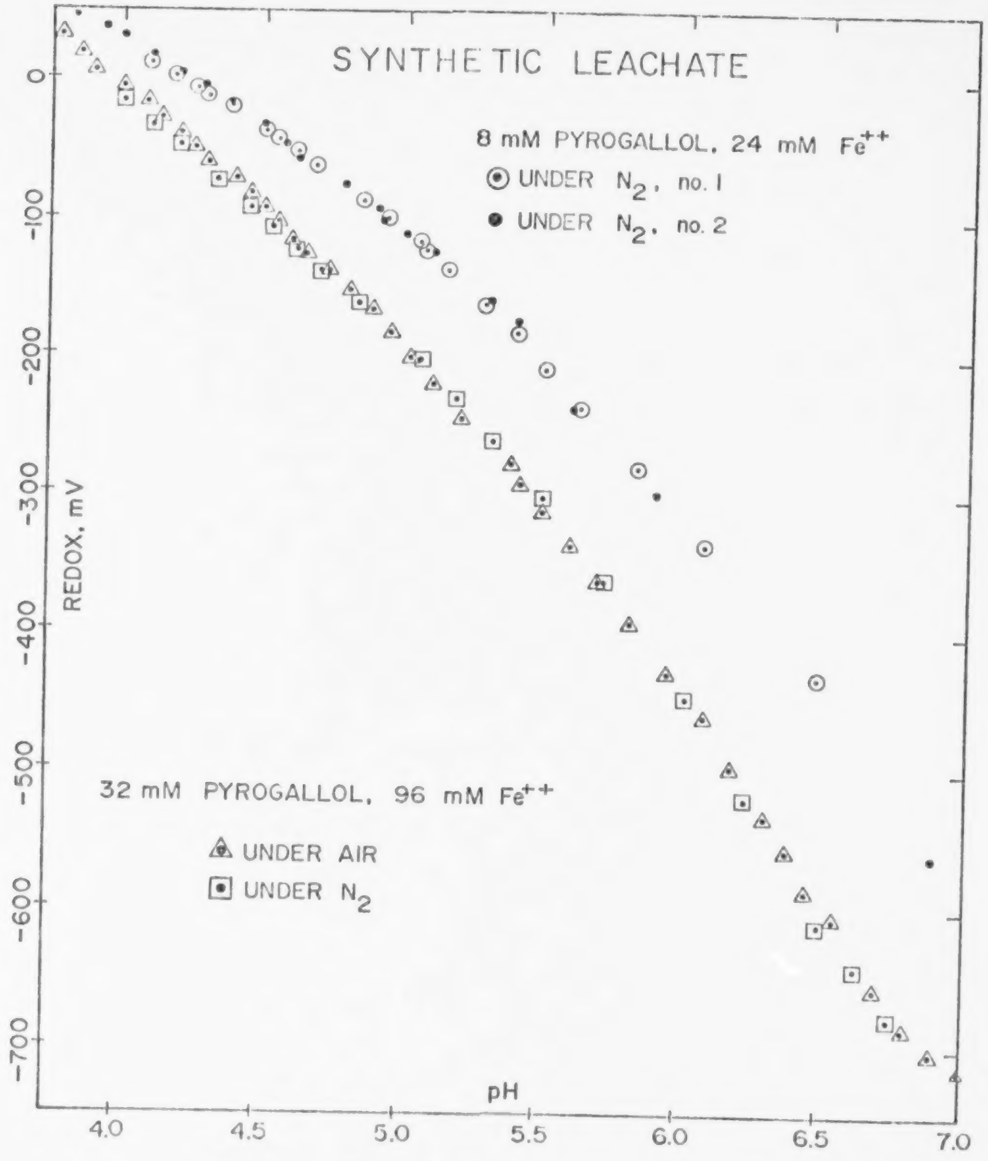

Figure 14. Change in synthetic leachate redox potential with pH.

Redox potential modeling is both one of the more difficult aspects of leachate to model and one that generates a lot of interest. The difficulty comes from tying to model a biological process chemically and trying to use as simple a procedure as possible.

—organic solvents

Only limited data is available on the ability of leachate to solubilize water-immiscible organic compounds. Burrows et al (14), Robertson et al (15), and Khare et al (16) have identified a number of organic constituents in leachate of interest for modeling the organic solubilizing capability. A summary of their data is given in Table 11. Unfortunately, none of these studies is useful for this modeling since none relate the various concentrations to landfill aging processes. However, identification of a number of organic solvents in leachate indicates the need for further research in order to determine the origins and concentrations of these materials in leachates, and their change in concentration as the landfill ages.

TABLE 11. ORGANIC COMPOUNDS OR CLASSES IDENTIFIED IN LANDFILL LEACHATE

Author	Compounds	Comments
Burrows & Row (14)	Acetone short chain alcohols short chain acids	leachate COD was 170,000 mg/ℓ, a very high value
Khare & Dundero (16)	Alkanes Ketones (Acetone, 2-butanone) $CHCl_3$, CCl_4 Aromatic solvents (benzene, toluene, xylene) short chain alcohols short chain amines short chain acids	Total carbon in leachate was 767 mg/ℓ, a relatively low value
Robertson, Toussaint & Jorqne (15)	phthalate esters Aromatic solvents (cresol, xylene) toluo ethyl-toluene sulfonamide alcohols methylpyridine ethers short chain acids	very small percentage of leachate T.O.C. identified

The majority of the known organic constituents in young leachate have been modeled already in the other sections. Thus, the synthetic leachate should provide a reasonable model for the organic solubilizing capability of leachate until further data becomes available.

—ionic strength

Ionic strength may affect the leaching of materials in three ways: by increasing the solubility through lowered activity coefficients, by ion exchange processes replacing an ion bound to an ion exchange site with one of the more predominant ions in solution, and by decreasing the size of the double layer around colloidal particles and promoting coagulation.

The ionic strength of leachate can be very high, but it is basically the result of only a few major species: Cl^-, HCO_3^-, SO_4^{2-}, and volatile acid anions, and Na^+, K^+, NH_4^+, Ca^{2+}, Mg^{2+}, and Fe^{2+} cations. Values for the maximum concentrations of these species reported in the literature are given in Table 12.

Unfortunately, charge balances for these major ions for some of the leachates reported in the literature indicate major charge inbalance; i.e., more plus charges than minus charges or vice versa. These calculations are shown in Table 13. While perfect charge balance would not be expected due to analytical errors in measuring the many parameters used, the magnitude of the difference in positive and negative charges indicates that many ions, particularly anions, are not being accounted for. Possibly some of the differences are due to (a) ionized organic hydroxyl groups, (b) association of some of the cations (particularly Fe) with OH groups, (c) incorrect estimations of the percentage of the volatile acid anions which are ionized, and (d) estimation of the HCO_3 concentration from the bicarbonate alkalinity since some of the volatile acids are also measured in the test for alkalinity. Whatever the reasons for the differences, it is important to keep in mind that knowledge of the composition of leachate, both inorganic and organic, is incomplete. Our understanding of the forms of the various parameters in the leachate is also, of course, very hazy. Chian et al (10) data presented in Table 13 was used since these samples have been analyzed thoroughly for both organic and inorganic constituents. Calculations using data from other researchers, Emcon Associates (5) and Pohland (7), also show major charge inbalances, with some positive and some negative residuals.

In modeling ionic strength, both sodium acetate and ferrous sulfate salts are used. The sodium, 150 mM, accounts for the ionic strength of both sodium and potassium ions. The iron, 24 mM, models the divalent cations. Its concentration is controlled by redox considerations and, so, is somewhat lower than the concentration which would be used if ionic strength alone dictated the concentration. The concentration of the primary anion present, acetate, is controlled by the buffering considerations for pH. Sulfate is added as the counter ion for iron.

TABLE 12. CONCENTRATIONS OF THE COMMON INORGANIC IONS FOUND IN LEACHATE

Source		Na	K	Ca	Mg	Cℓ	SO_4
		mM					
Emcon Assoc. (5,6)	A	39.1	23.4	56.0	43.6	117	2.6
	B	74.3	41.2	74.8	37.0	113	16.5
	C	41.3	21.6	42.4	44.0	36.6	9.2
Calif. Lab*		78.5	47.6	64.3	16.9	66.2	7.6
Calif. Fld*		30.9	4.9	8.9	4.5	24.4	7.3
Qasim et al (9)		62.6	96.4	101.7	17.3	65.1	10.4
Fungaroli (8)		165				56.3	4.2
Pohland (7), Fill 1		7.8		31.3	10.7	10.8	1.6
Hughes et al (13)		52.2	20.2	7.5	12.3	43.3	8.3
Stegmann (17)		109.3	61.4	8.4	49.4	106.6	14.6
Merz[†]		78.5	47.6	64.1	16.9	66.2	
Meichtry[†]		33.3	1.7	180	642	18.6	12.4
U. IL[†]		59.1	29.2	93.5	26.7	41.7	11.6
Boon Co.[†]		32.6	22.0	58.0	21.8	59.0	9.5
Mad. MUNC[†]		68.7	58.8	97.5	46.9	69.5	16.2
Mad. UMC[†]		14.3	23.0	14.3	9.1	13.4	0.8
Value which could be used		100	50	100	50	100	15

[*] From Emcon Associates (6).

[†] From Chian et al (10).

TABLE 13. CHARGE BALANCE CALCULATIONS FOR LEACHATE DATA GIVEN BY CHIAN ET AL (10)

	U.I. 2/1	MUNC	UMC	BOON
	mequiv/liter			
Positive Ions				
Na^+	59.1	68.7	14.3	32.6
K^+	29.2	58.8	23.0	22.0
NH_4^+	28	73.4	20.4	17.7
Ca^{++}	187	195	28.6	116.0
Mg^{++}	53.4	93.8	18.2	43.6
Zn^{++}	3.2	11.4	0.0	1.6
Fe^{++}	78.8	37.4	3.2	26.4
Total Positive Charge	438.7	538.5	107.7	259.9
Negative Ions				
Volatile Acids				
-Completely Ionized*	(199.5)	(300.1)	(25.7)	(73.4)
-Using pK†	159.6	257.1	20.6	55.1
HCO^-_3‡	5.7	4.6	2.8	1.8
$SO_4^=$	23.2	32.4	1.6	19.0
Cl^-	41.7	69.5	13.4	59.0
NO_3^-	0.0	0.0	0.0	0.7
$H_2PO_4^-$	0.2	0.9	2.7	0.0
Total Negative Charge	231.4	364.5	41.1	135.6
Apparent Net Charge	207.3+	174.0+	66.6+	124.3+

*Assuming all measured volatile acids are ionized. Value not used in calculation.

†Assuming partly ionized, in accordance with pKa values.

‡Assuming all bicarbonate alkalinity as HCO_3^-.

The complete composition and directions for the synthetic leachate are given in Table 14. The Fe-Pyrogallol complex will oxidize and precipitate if exposed to air for very long. Therefore, experimental vessels need to be either filled almost completely or purged with N_2 before sealing. Containers should be sealed tightly during experiments. For long term experiments, glass containers should be used to avoid diffusion of oxygen through the walls of the container. If kept from oxidizing, the solution is stable for several weeks.

TABLE 14. DIRECTIONS FOR PREPARING THE SYNTHETIC LEACHATE

Composition:

0.15 M sodium acetate
0.15 M acetic acid
0.050 M glycine
0.008 M pyrogallol (1,2,3-trihydroxybenzene)
0.024 M ferrous sulfate

Directions:

I. Concentrated acetate buffer - glycine solution (10X)

Dissolve 204.1 gm $NaC_2H_3O_2 \cdot 3H_2O$ and 37.5 gm glycine in approximately 500 ml deaerated distilled water in a 1 liter volumetric. Add 86.2 ml glacial acetic acid. Bring to room temp, then dilute to volume. Solution is stable for at least a week (probably more).

II. SL-A solution (synthetic leachate without the Fe)

Place 100 ml acetate-glycine solution in a 1 liter volumetric. Add 1.05 gm pyrogallol and approximately 500 ml deaerated water. After pyrogallol dissolves, bring to volume with deaerated water. Prepare fresh daily and store in a stoppered bottle or volumetric flask.

III. Synthetic leachate

Add 200 ml of SL-A solution to weighed waste in appropriate flask. Add 1.38 gm $FeSO_4 \cdot 7H_2O$ to solution, seal and shake. If an amount of synthetic leachate other than 200 ml is required, use 0.69 gm $FeSO_4 \cdot 7H_2O$ per 100 ml.

---

(5) Limitations of the synthetic leachate

There are two known limitations of the synthetic leachate (SL) as presently formulated. First, the synthetic leachate is based on the data currently available, this data being limited or nonexistent in some areas. For example, data regarding organic solvents in leachate is very limited. Many solvents have been found, as discussed previously, but neither their concentrations nor their dynamics in the life cycle of the landfill have been investigated. As another example, one might expect some organic sulfur compounds to occur in leachate and to be very strong complexing agents, but these compounds have not been measured in leachate and, so, cannot be modeled. Second, the synthetic leachate does not model the biota or particulate matter found in leachate. It is impossible to model the diverse effects of the bacteria chemically, and to include bacteria in the synthetic leachate involves a much more complicated and unstable leachate than was desired. Materials that will leach from the waste in a landfill through sorption onto particulate matter will not be leached by the synthetic leachate.

(6) A non-anaerobic modified synthetic leachate

The anaerobic nature of the synthetic leachate necessitates careful handling of the leachate to minimize air contact. Oxidation of the leachate results in the possibility of errors due to coprecipitation and adsorption of the measured species on the precipitate. To avoid these problems, a non-anaerobic synthetic leachate has been designed. Salicylic acid (2-hydroxybenzoic acid) is used instead of pyrogallol as a complexing agent, with the C content in the leachate from the complexing agent at 0.05 M. Ferrous sulfate is left out. Otherwise the leachate remains the same. The composition of the non-anaerobic leachate is as follows:

0.15 M sodium acetate
0.15 M acetic acid
0.050 M glycine
0.007 M salicylic acid.

This is not as complete a model of municipal landfill leachate as the anaerobic synthetic leachate, since it does not model a significant feature of municipal landfill leachate. It is, however, much easier to use. The modified leachate has not been extensively tested, as has the synthetic leachate, and is offered only as a suggested non-anaerobic synthetic leachate.

(7) Concluding statement

A synthetic leachate has been produced which is modeled after a very aggressive municipal refuse landfill leachate. In order to verify the leaching aggressiveness of the synthetic leachate towards industrial wastes, it is necessary to use aggressive real leachate as a comparison. Verification could be accomplished best by comparing the behavior of a waste leached with the synthetic leachate with the behavior of the same waste in an actively decomposing landfill.

The synthetic leachate provides a first generation model of aggressive municipal leachate for use in a leaching test. Using the synthetic leachate makes it possible to predict the waste constituents which would dissolve in aggressive municipal leachate and, further, to estimate the constituent concentrations which might be found in the real leachate. Note that if the waste being tested is to be co-disposed with a biologically decomposable industrial waste, the synthetic leachate or a variation of it may be the best leaching media to simulate landfill conditions

Solid-Liquid Ratio

A. General Considerations

The solid-liquid ratio (waste/leachate) is very important in the design of a standard leaching test. The ratio can determine whether or not saturated conditions are reached and, if not, can affect the concentrations of the unsaturated parameters.

Two practical considerations are of importance. If the solid-liquid ratio is very low, sampling and analytical errors will be magnified when the release rate is normalized per gram of waste. Furthermore, the concentrations of some interesting trace contaminants may fall below detection limits. At the other extreme, a very high ratio can lead to difficulties in stirring or separation techniques and can take a long time to stabilize. Since landfills have high solid-liquid ratios over the short term, the values chosen for testing were on the high solid-liquid ratio end of the spectrum when choices were necessary. In most existing leaching tests the solid-liquid ratio is based on waste weight and liquid volume or waste surface area-liquid volume ratios could be used. Surface area is waste specific.

The solid-liquid ratio influences the results from a leaching test and so should be standard for all wastes. If the solid-liquid ratio is based on weight, the surface area-liquid ratio then depends on the specific weight of the waste. For two wastes with the same particle size but very different specific gravities, e.g., lead and styrofoam, the surface area of the lighter waste would be much greater in the test if the solid-liquid ratio is based on weight. If the elution rate is dependent on surface area, it would be more realistic to keep the waste surface area-leachate volume ratio constant rather than the waste weight-leachate volume ratio. Leaching will be controlled by surface area when desorption and diffusion are the major factors controlling release, and over the short term when solubility is the controlling factor. It is very difficult to determine when leaching is controlled by surfaces for different wastes. Also the surface area of a waste (e.g., styrofoam) may change when the waste is landfilled, compressed by the weight of overlaying wastes and decomposes. Original densities of the solid portions of the different wastes used during the project are estimated to have ranged from 0.8 to 2.5 gr/cm^3. Therefore, the change in the solid-liquid ratio, based on weight, would not have been expected to vary over a factor of 2.5, if the surface area-leachate volume ratio was kept constant. Unfortunately, it is very difficult to determine the surface area of different kinds of wastes, especially if these wastes are sludges, oils, or mixtures of different components.

In summary, the effect of using a constant waste surface area-leachate ratio in a test rather than using a constant waste weight-leachate ratio is difficult to predict, and to apply to a full size landfill. Determination of surface area is difficult for many wastes. Furthermore, for viscous liquids which are non-filterable and therefore treated as solids in the test, surface area will depend on the shape of the container. Therefore, in this project, solid-liquid ratios are always based on weight.

In addition to simple solid-liquid ratios, two different landfill situations were modeled and tested. These two situations, shown in Figure 15, are as follows:

1. The landfill contains a very thick layer of waste, such that a drop of liquid traveling through the waste will come in contact with a large amount of waste (lower drawing).
2. The landfill contains only a thin layer of waste, such that a drop of liquid traveling through the waste comes in contact with only a small amount of waste (upper drawing).

Tests simulating Situation 1 will give values indicative of the maximum concentration, while tests modeling Situation 2 will give an indication of the maximum release of leachable matter from a unit amount of waste.

B. Experimental Results and Discussion

Test Series V and R addressed the solid-liquid ratio effects and modeled the different landfill situations described above. In Series V distilled water was used as the leaching medium, with intermittent shaking (further defined in the next section) as an agitation method. Each elution lasted 24 hours, was run at room temperature, and was followed by 0.45 micron filtration to separate the leachate and solids. The different procedures within Series V are shown in Figure 16. Procedures (tests) 6, 7, and 8 studied directly the effects of the solid-liquid ratio. Three separate replicate flasks were set up at a given ratio, with one flask removed for analysis daily. Three ratios based on weight were tested, namely 1:10 (#6), 1:7 (#7), and 1:4.8 (#8).

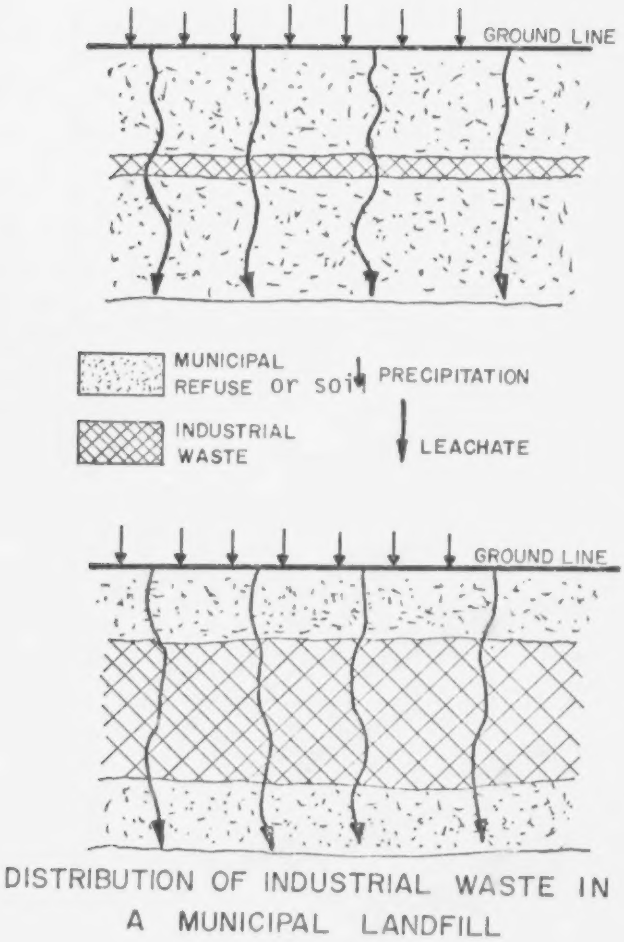
GROUND LINE
MUNICIPAL REFUSE or soil
PRECIPITATION
INDUSTRIAL WASTE
LEACHATE
GROUND LINE
DISTRIBUTION OF INDUSTRIAL WASTE IN A MUNICIPAL LANDFILL

Figure 15. Landfill situations modelled in series V, procedures 1 and 5.

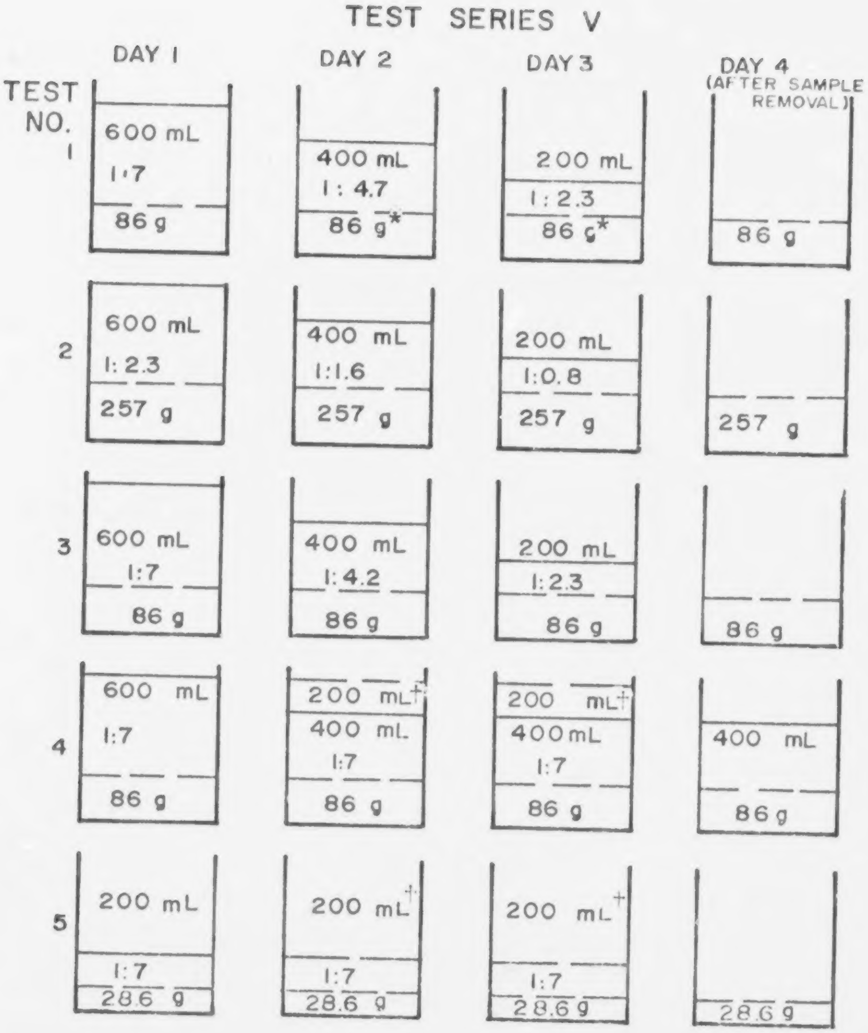
TEST SERIES V
DAY 1
DAY 2
DAY 3
DAY 4 (AFTER SAMPLE REMOVAL)
TEST NO.
1
600 mL
1:7
86 g
400 mL
1: 4.7
86 g*
200 mL
1: 2.3
86 g*
86 g
2
600 mL
1: 2.3
257 g
400 mL
1:1.6
257 g
200 mL
1:0.8
257 g
257 g
3
600 mL
1:7
86 g
400 mL
1:4.2
86 g
200 mL
1:2.3
86 g
86 g
4
600 mL
1:7
86 g
200 mL†
400 mL
1:7
86 g
200 mL†
400 mL
1:7
86 g
400 mL
86 g
5
200 mL
1:7
28.6 g
200 mL†
1:7
28.6 g
200 mL†
1:7
28.6 g
28.6 g

Figure 16. A diagram of test series V.

*Fresh solids.

†Fresh eluent.

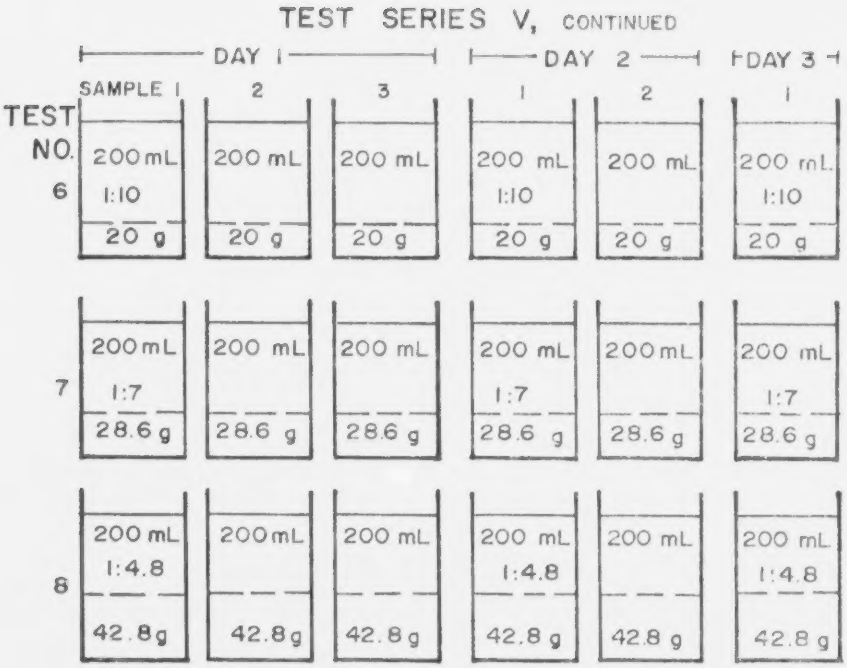
TEST SERIES V, CONTINUED
DAY 1
DAY 2
DAY 3
SAMPLE 1
2
3
1
2
1
TEST NO.
6
200 mL
1:10
20 g
7
200 mL
1:7
28.6 g
8
200 mL
1:4.8
42.8 g

Figure 16. A diagram of test series V (continued).

Landfill Situation 1, described above, was modeled in procedure number 1. Initially, 86 gm of waste were mixed with 600 ml of eluent (1:7 ratio). After 24 hours, the mixture was filtered, 200 ml of the filtrate removed for analysis and the remaining 400 ml were mixed with 86 gm of fresh waste (1:4.7 ratio). The flask was again intermittently shaken for 24 hours and the separation process repeated. 200 ml of the filtrate were removed for analysis and the final 200 ml mixed with 86 gm of fresh waste (1:2.3 ratio). The mixture was intermittently shaken 24 hours, then the final filtration was made. A total of 257 gm of waste were used.

Procedures 2 and 3 were slight modifications of procedure 1. In both cases, the filtered eluent was mixed with the same waste with which it was initially contacted rather than replacing the waste each day. 257 gm of the waste were used in procedure 2, while in procedure 3, 86 gm were used.

Procedure 5 models landfill Situation 2 described above. Initially, 200 ml of distilled water were mixed with 28.6 gm of waste (1:7 ratio) and intermittently shaken. After 24 hours, the mixture was filtered, the filtrate analyzed and the solids returned to the flask. 200 ml of fresh eluent were added and the mixture again intermittently shaken for 24 hours. After filtration, 200 ml of fresh eluent were again added to the same waste sample. After 24 more hours, the sample was filtered and the experiment ended. This procedure was determined to be the best for foundry sand (Kunes, et al. 13).

Procedure 4 is a cross between 3 and 5. It is similar to procedure 3, except that the 200 ml of solution removed for analysis were replaced with fresh eluent, thus maintaining a constant solid-liquid ratio.

Representative results for Series V are presented in Figures 17, 18, and 19 in which paint waste (AA) was used. COD results are given both as concentration and as the amount leached per unit mass of waste. As can be seen from the results presented, the various procedures (tests) used may influence the results. Redox and pH are particularly interesting parameters, since changes in these parameters may affect the leaching of other parameters. The choice of procedure may have an important effect on the leaching characteristics of the waste.

The results for COD (Figure 18) and Conductivity (Figure 19) illustrate the information obtainable from the different tests. In procedure 1, the COD concentration increases linearly, indicating unsaturated conditions in the leachate. Procedure 5 shows that the COD is not completely leached from the waste on the first day, since subsequent elutions also have significant COD concentrations. Normalizing the COD release on a per weight basis shows that in procedure 1, adding fresh waste does not change the COD released per kilogram of waste, while the successive elutions in procedure 5 increase the release per kilogram of waste. Procedures designed to give maximum concentrations (#1) and maximum release (#5) are both important in a standard leaching test; unfortunately, one test cannot give both. Data on the maximum concentrations and maximum releases after three days for the various wastes tested in Series V, including the paint waste, are given in Table 15.

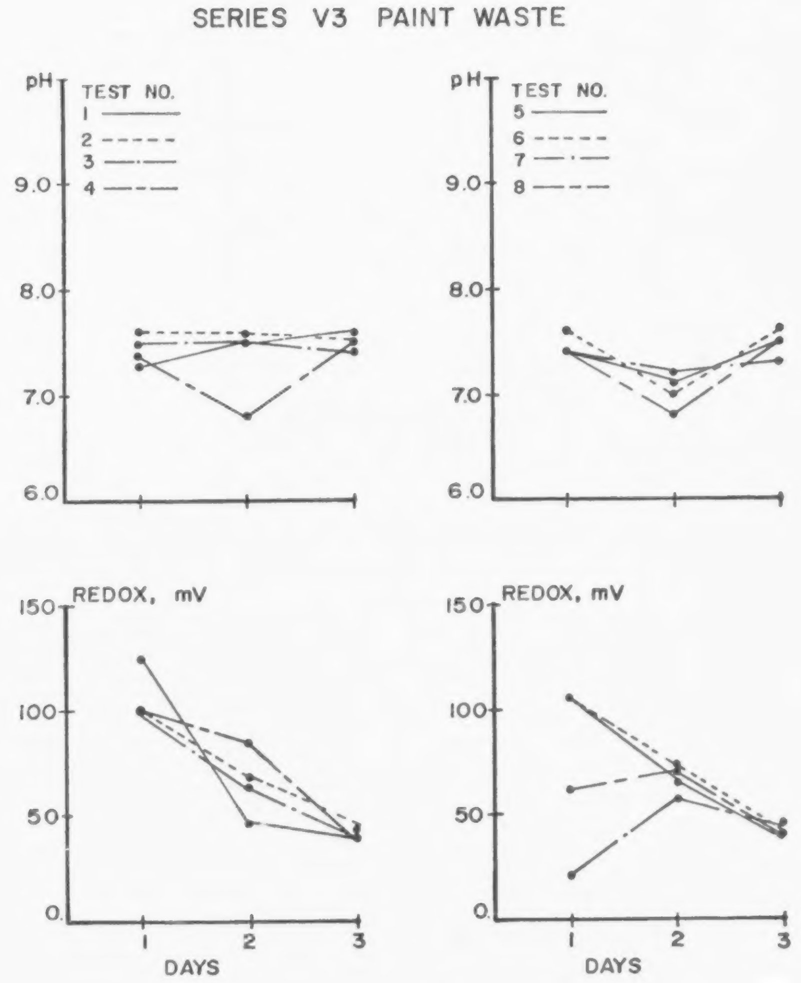

Figure 17. pH and redox results from series V using paint waste.

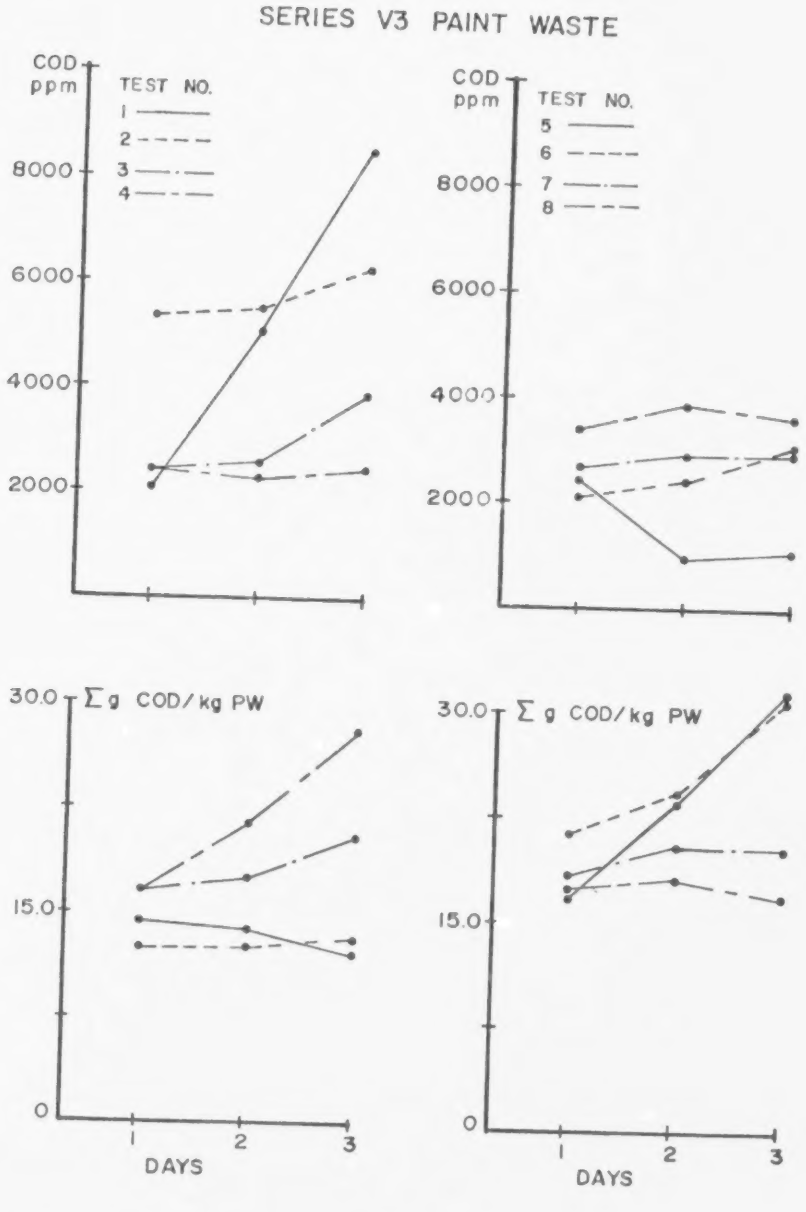

Figure 18. COD results from series V using paint waste.

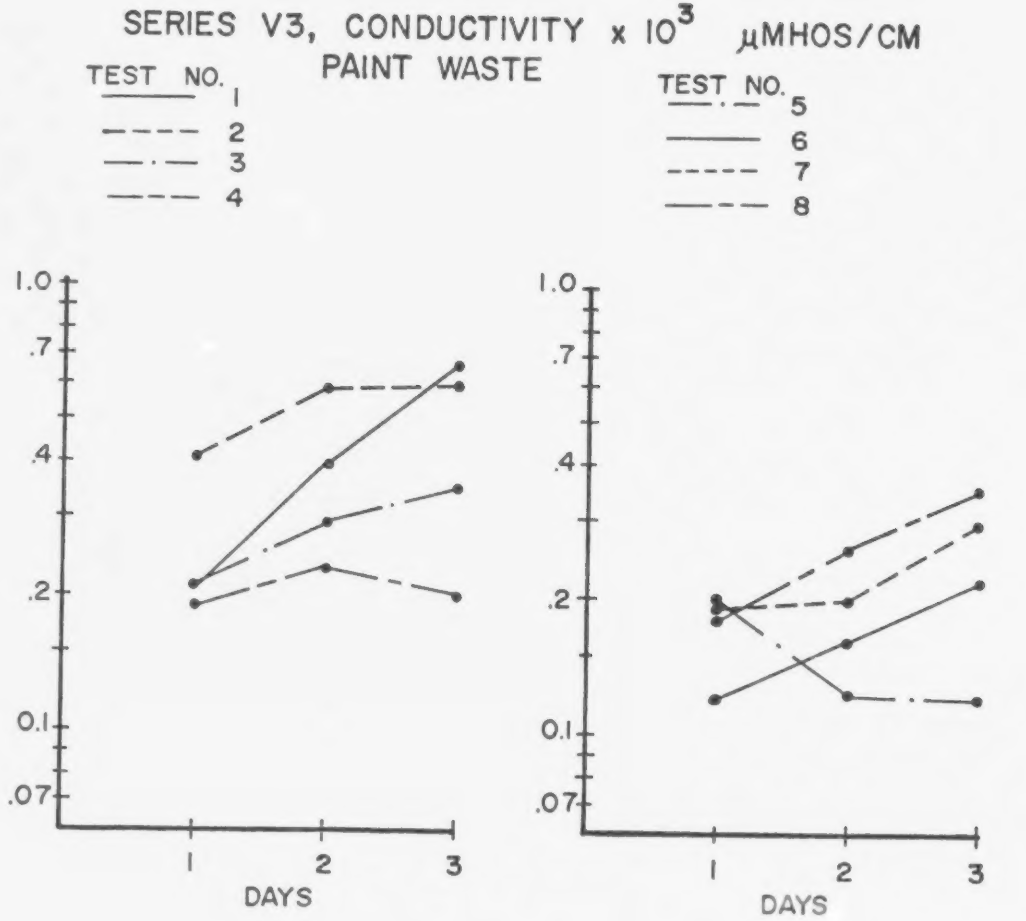

Figure 19. Specific conductance results from series V using paint waste.

TABLE 15. MAXIMUM CONCENTRATIONS AND RELEASE

1.

Waste	Test Procedure	COD [ppm]	Conduct. × 10^3 [μmhos/cm]	K [ppm]	Ca [ppm]	Fe [ppm]	Mg [ppm]	Zn [ppm]
Fly Ash (EPA)	1	61	0.82	17	-----	-----	-----	-----
	5	7	0.21	0.5	-----	-----	-----	-----
Paper Mill Sludge (N)	1	1012	1.41	6.6	4.3	0.37	150	-----
	5	80	0.11	2.0	1.3	0.12	8	-----
Municipal Refuse (City of Madison)	1	2990	3.4	190	116	11.0	58	5.7
	5	160	0.135	2.8	5.5	0.2	3	0.32
Paint Waste (AA)	1	8500	0.67	-----	-----	-----	-----	0.84
	5	1090	0.12	-----	-----	-----	-----	0.19
Fly Ash	1	875*	6.6	108	350	-----	-----	-----
	5	34*	0.3	3.5	25	-----	-----	-----

*After two days

TABLE 15 (continued)

2.

Waste		COD [mg/kg]	K [mg/kg]	Ca [mg/kg]	Fe [mg/kg]	Mg [mg/kg]	Zn [mg/kg]
Fly Ash (EPA)	1	105	18.4	-----	-----	-----	-----
	5	259	21.7	-----	-----	-----	-----
Paper Mill Sludge (N)	1	1200	13	7.3	0.5	157	-----
	5	4000	51.3	27.4	2.6	238	-----
Municipal Refuse (City of Madison)	1	5040	289	175	14.3	87	10.8
	5	11190	513	296	15.4	153	39.8
Paint Waste (AA)	1	12100	-----	-----	-----	-----	1.5
	5	31500	-----	-----	-----	-----	5.1
Fly Ash	1	2380	138	671	-----	-----	-----
	5	3000	199	1855	-----	-----	-----

As might be expected, the COD concentration increases as the solid-liquid ratio increases (#6 < #7 < #8), while the release per kg decreases with increasing solid-liquid ratio.

In the following figures, the results from test series V are summarized by chemical parameters (Figure 20—COD, Figure 21—K, Figure 22—Fe, Figure 23—Zn, Figure 24—Mg, Figure 25—Cu). The different curves on each graph relate to the different wastes used:

#1	Fly Ash (EPA)
#2	Papermill Sludge (N)
#3	Paint Waste (AA)
#4	Fly Ash
#5	Shredded Municipal Waste (City of Madison)

The upper two graphs on each figure relate to the concentrations of the various parameters, and the lower two graphs to the release of each parameter per weight of waste. The two graphs on the left summarize the results after one day for different solid-liquid ratios, and the two graphs on the right for test procedures 1 and 5 after three days. These are the two procedures suggested to best indicate maximum concentration (procedure 1) and release (procedure 5), and will be termed Procedures C and R, respectively, for the remainder of this report.

These results suggest that all six chemical parameters measured for the five wastes are influenced in a similar manner by variations in the solid-liquid ratio. Concentrations decrease with decreasing solid-liquid ratio while the quantity of release increases. The differences in results from test procedures #1 and #5 are noteworthy. This summary shows that it is necessary to determine a standard solid-liquid ratio for a standard leaching test. It also shows that the results from procedures 1 (C) and 5 (R) are very different, and that determination of both the maximum concentration and maximum release are important in understanding the leaching characteristics of a specific waste.

Tests (procedures) 1, 3, 4, 5, and 7 in test series V all have solid-liquid ratios of 1:7 on the first day, and concentrations would be expected to be very similar for a given waste. Table 16 provides a statistical evaluation of the test series V data, which shows that the deviations from the means are generally small compared with the means, indicating fairly good reproducibility. Some results are approaching accurate detection limits for the analytical procedures used, and do not relate so much to the reproducibility of the test procedure itself.

After performing these preliminary tests on the influence of the solid-liquid ratio upon the leachability of wastes, two more test series (R1 and R2) were designed. In the first of these (R1), the range of solid-liquid ratios was larger than in series V (1:4, 1:2, 1:10, 1:20, 1:100), with waste weight measured as dry weight of the solids. This was necessary because the moisture content after the solid-liquid-separation step will vary depending on the waste characteristics. For example, a waste that has 90% moisture content will subject a smaller amount of solids to the leaching media than a waste that has a 50% moisture content, as shown in the following example.

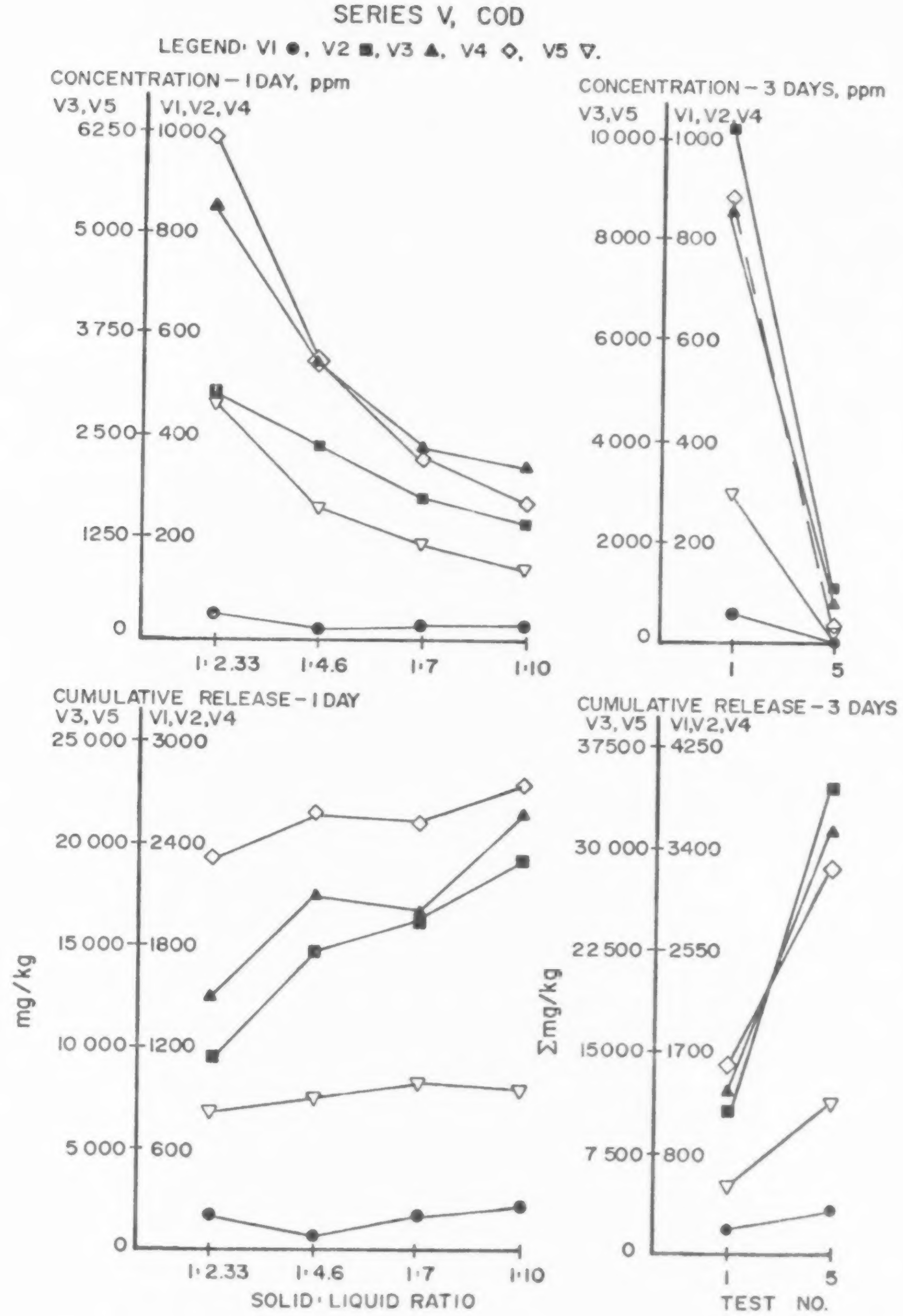

Figure 20. Summary of results from series V: COD.

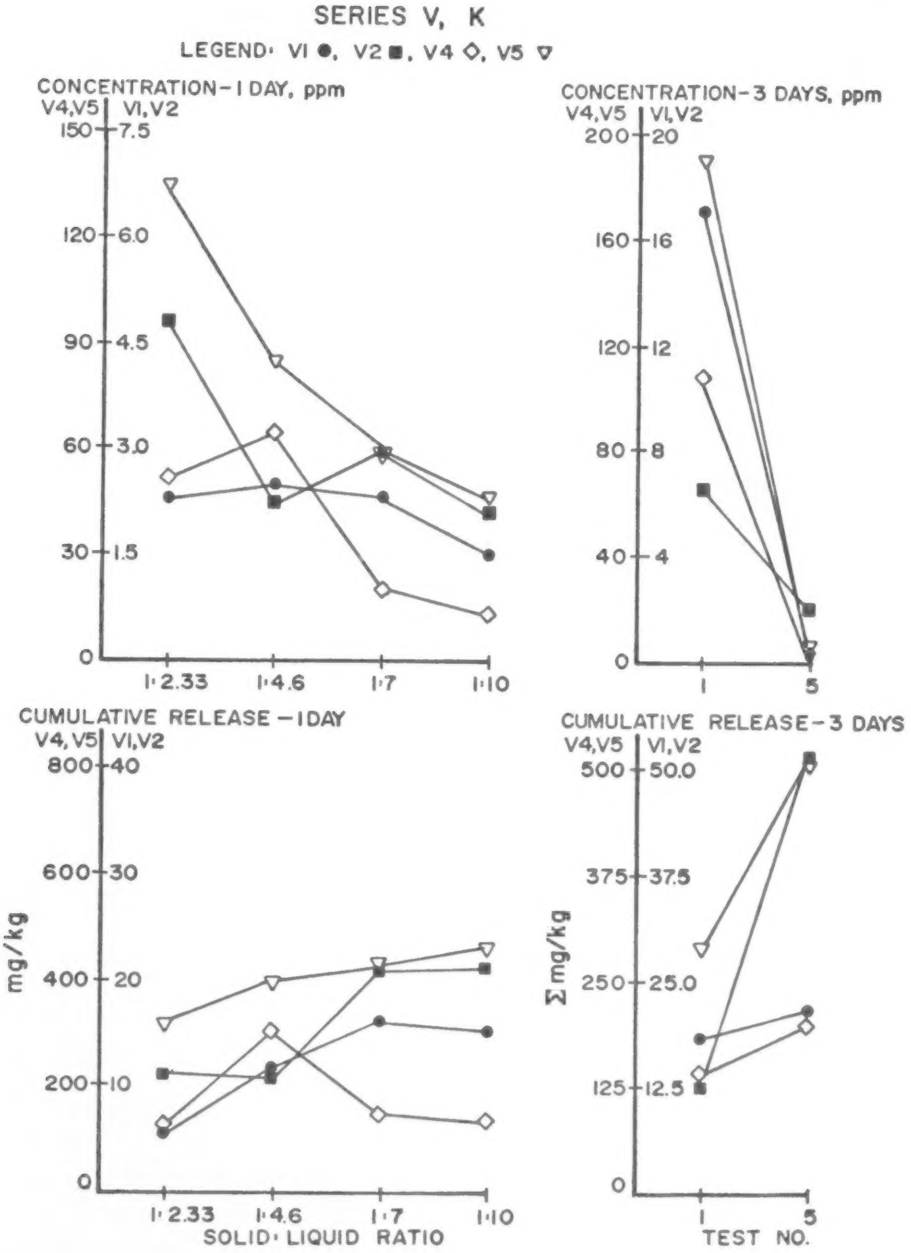

Figure 21. Summary of results from series V: K.

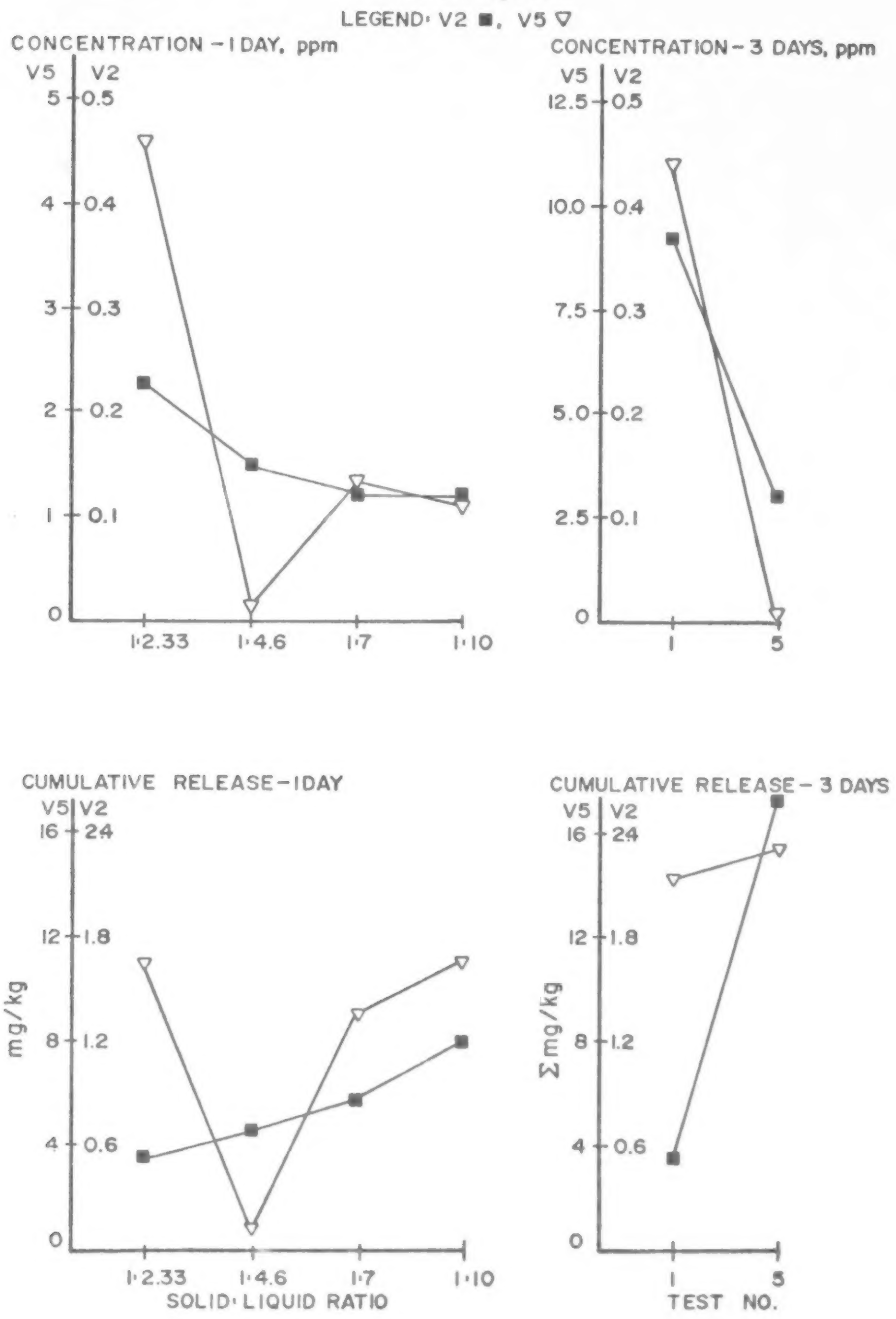

Figure 22. Summary of results from series V: Fe.

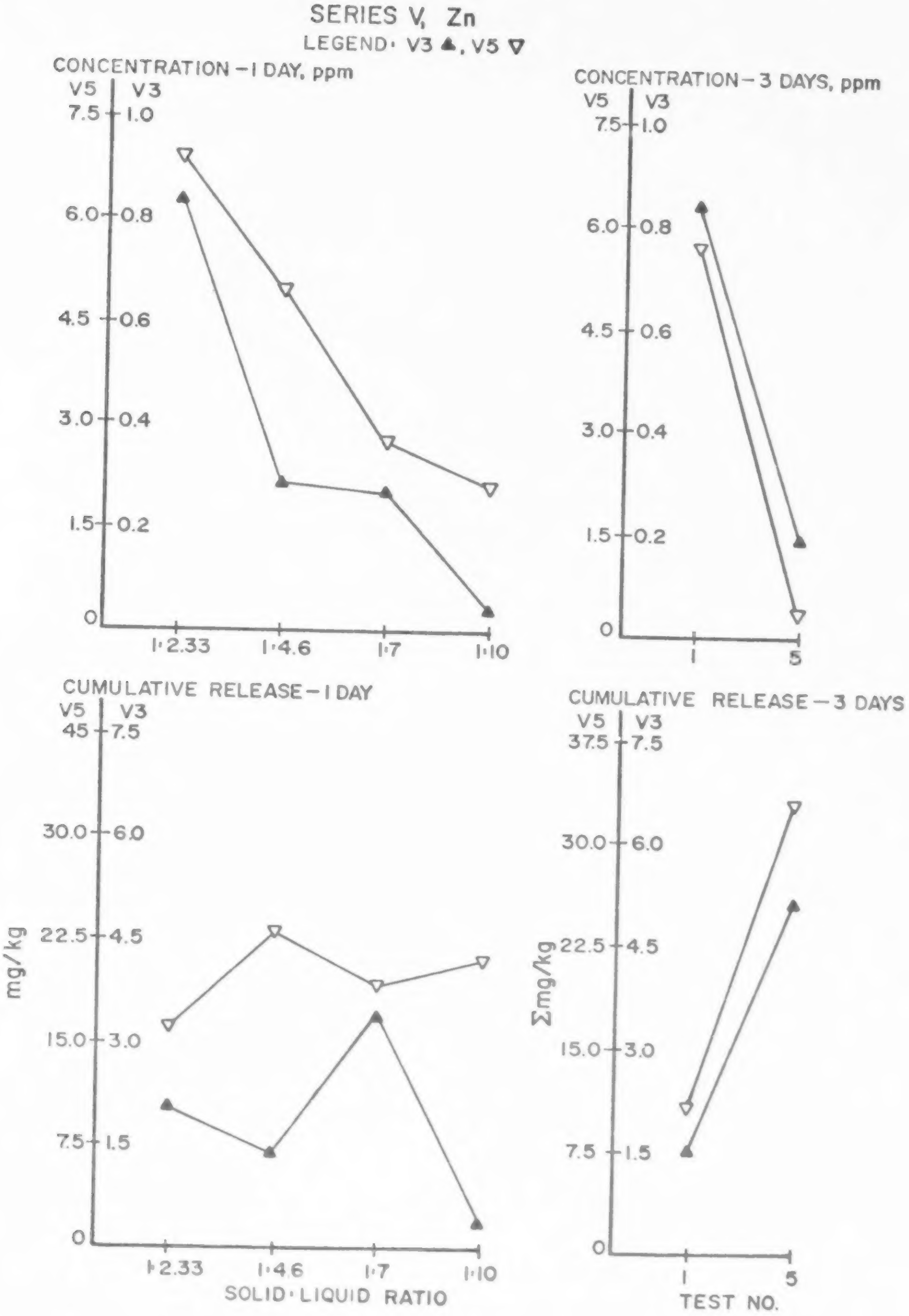

Figure 23. Summary of results from series V: Zn.

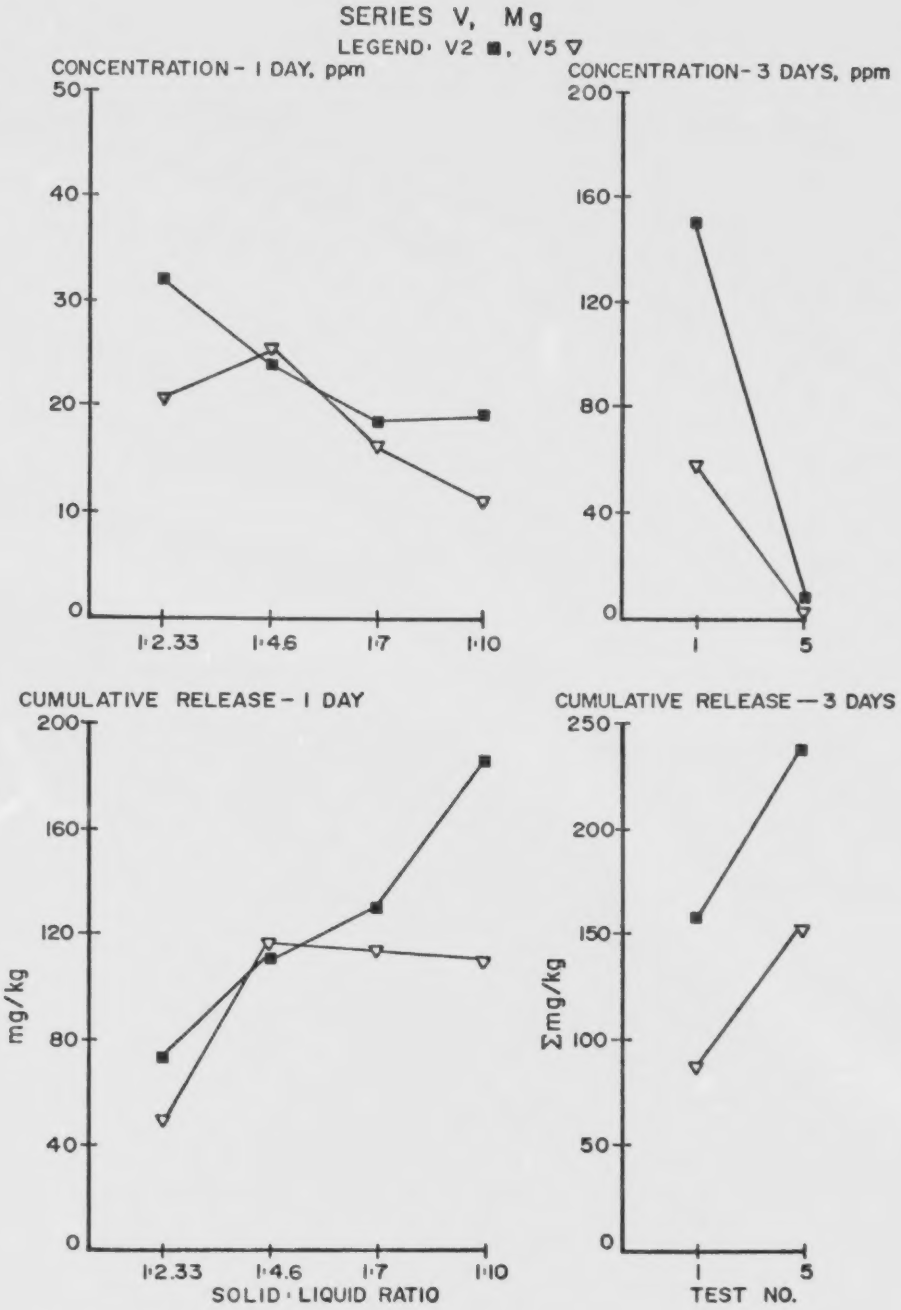

Figure 24. Summary of results from series V: Mg.

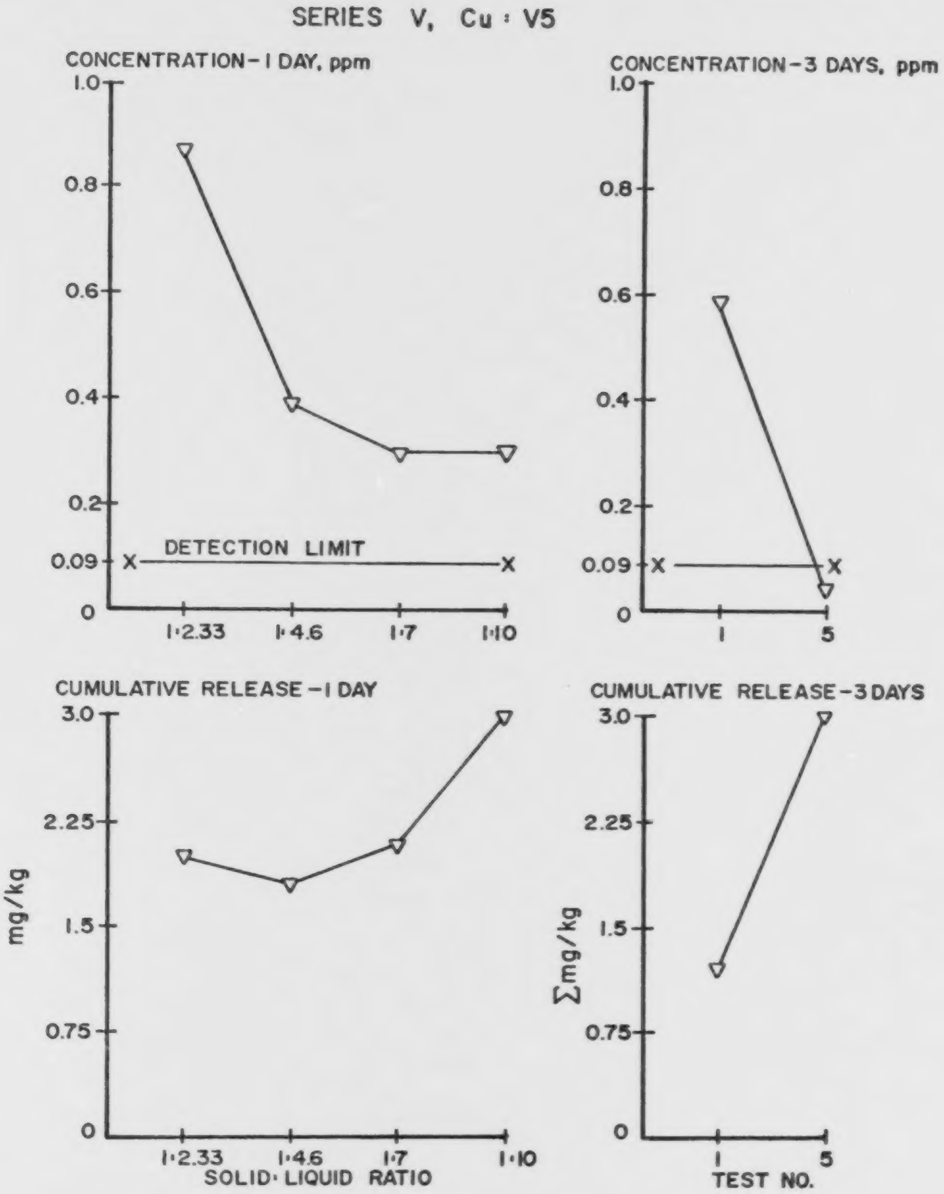

Figure 25. Summary of results from series V: Cu.

TABLE 16. STATISTICAL RESULTS FOR TEST SERIES V FOR SOLID-LIQUID RATIO OF 1:7 (WET WEIGHT) AFTER ONE DAY

Eluent	Waste*	No. Samples	$\bar{X}$	S	S%	$\bar{X} \pm 2S$†
Mg	2	5	18.6	2.3	±12	14, 23.2
	5	5	23.2	14.9	±64	-6.6, 53
Zn	3	5	0.274	0.025	± 9	0.22, 0.32
	5	5	2.76	0.47	±17	1.8, 3.7
Cu	5	5	0.298	0.047	±16	0.2, 0.39
Fe	2	5	0.124	0.0134	±11	0.097, 0.15
	5	5	1.3	0.276	±21	0.75, 1.85
K	1	4	2.3	1.0	±43	0.3, 4.3
	2	5	2.97	1.95	±66	-0.9, 6.9
	4	5	20.4	1.69	± 8	17, 23.8
	5	5	59.8	4.21	± 7	51.4, 68.2
COD	1	5	29.2	1.64	± 6	25.9, 32.5
	2	5	280	19.7	± 7	241, 320
	3	5	2390	192	± 7	2010, 2770
	4	5	362	42.8	±12	276, 448
	5	5	1200	39.8	± 3	1120, 1280

*Waste 1: Fly Ash (EPA)
2: Papermill Sludge (N)
3: Paint Waste (AA)
4: Fly Ash
5: Municipal Waste

$\bar{X}$ = mean

S = standard deviation

S% = standard deviation as a percentage of $\bar{X}$

†In the interval $\bar{X} \pm 2S$ are 95% of all values $(n = \infty)$, assuming values are normally distributed.

> Assume the solid-liquid ratio is based on wet weight and is 1:10. Also assume the spcific weight of the solids is 1 kg/l for two wastes, A and B, which differ in initial moisture content after filtration. The amount of dry solids would be:
>
> | Waste | Moisture Content | Wet weight of waste / 100 ml leaching solution | Dry weight of waste / 100 ml leaching solution |
> |---|---|---|---|
> | A | 90% | 10g | 1g |
> | B | 50% | 10g | 5g |
>
> Therefore, the amount of solids used with waste B is 5 times higher than the amount used with waste A.

Initially, it was assumed that all wet wastes could be dewatered to approximately the same moisture content, avoiding such problems. Experience has indicated that this is not the case, and that dry weight must be used.

Series R1 was performed with fly ash (particle size <0.5 mm) and paint waste (AA). The procedure used was #5(R) from series V, where the liquid is replaced every 24 hours to maximize release. Five elutions were used. 0.1 N H_2SO_4 and synthetic leachate were used as leaching media. The composition and other characteristics of the synthetic leachate were described in the previous chapter.

The influence of solid-liquid ratio on release after three days can be seen from Figures 26 and 27. As seen previously (series V), release generally is greater with lower solid-liquid ratios, although some parameters do not show much influence over the range of ratios 1:4, 1:7, 1:10, and 1:20. Based on results for the series R1, a 1:10 ratio based on dry weight appears reasonable. This ratio represents a compromise which generally provides concentrations above detection limits without the lower release of materials observed with the 1:7 and 1:4 ratios.

In the second test, series R2, the influence of the solid-liquid ratio was tested again, but this time with test procedure 1 (C) of series V, where the waste is replaced after each elution to maximize the concentrations of leached substances. Three different ratios were used: 1:5, 1:10, 1:20. Lower ratios were not tested because the objective of the test procedure was to achieve maximum concentration. Up to five elutions were made in test series R2. The amount of waste that was replaced each 24 hours was constant, but the leachate volume decreased because samples were extracted for analysis after each elution. Therefore, the solid-liquid ratio was not constant during the test. Table 17 shows the different solid-liquid ratios.

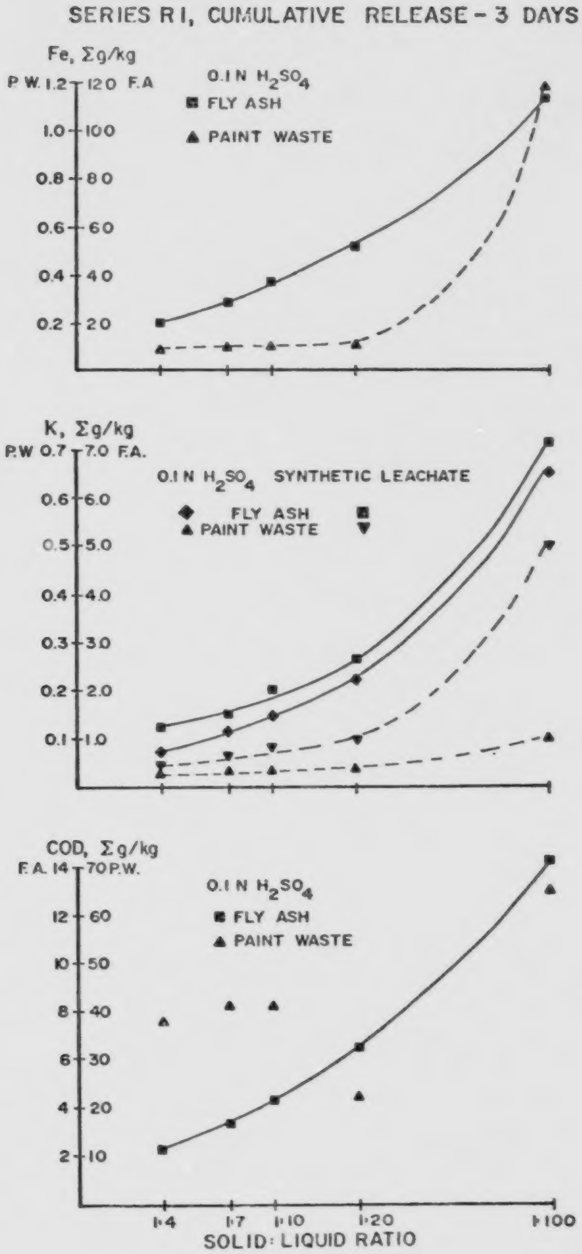

Figure 26. Effect of solid-liquid ratio on three-day Fe, K, and COD release.

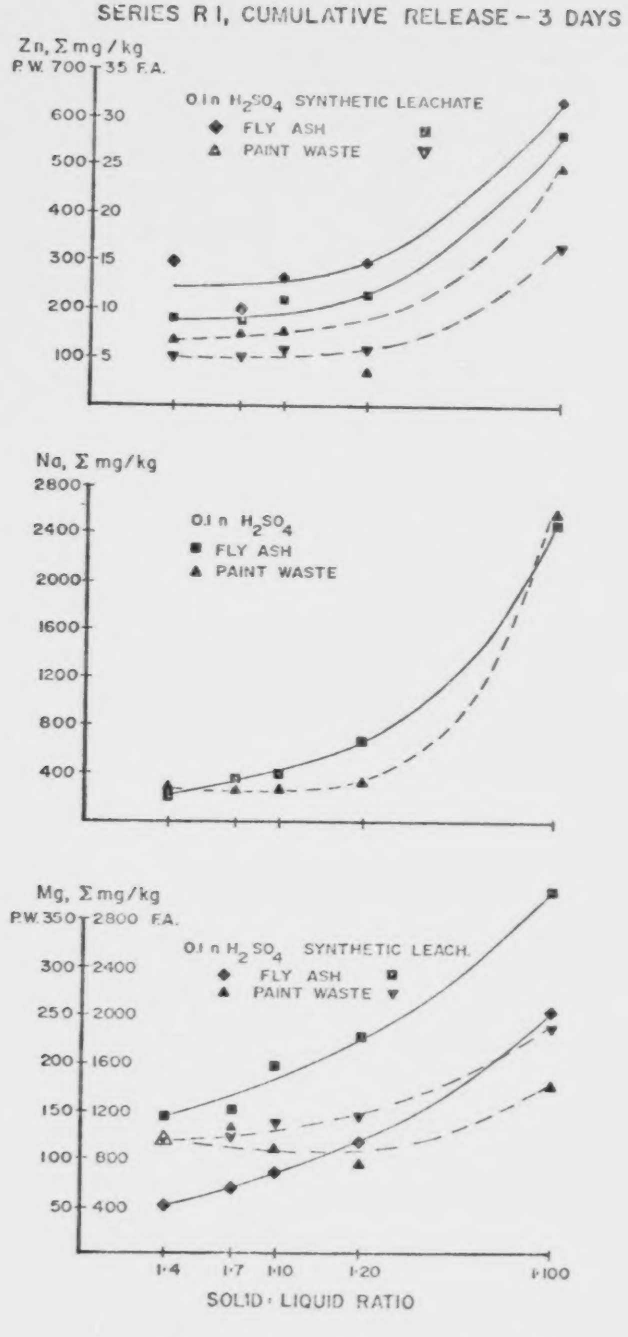

Figure 27. Effect of solid-liquid ratio on three-day Zn, Na, and Mg release.

TABLE 17. SOLID-LIQUID RATIOS DURING TEST SERIES R2 (BASED ON DRY WEIGHT)

Elution	Solid-liquid Ratios		
1	1:10	1:5	1:20
2	1:8.5	1:4.2	1:17
3	1:6.9	1:3.5	1:12.8
4	1:5.4	1:2.7	1:10.4
5	1:3.8	1:1.9	1:7.6

The tests were performed with 0.1 N H_2SO_4 and synthetic leachate using fly ash (particle size $<$ 0.5 mm) and paint waste (AA) as solids. In the case of fly ash the test could not be continued for 5 elutions because of leachate loss due to absorption by fly ash and other losses. In this case. when so many elutions were desired, fly ash should have been wetted to field capacity with additional amounts of 0.1 N H_2SO_4 or synthetic leachate.

pH and Na, Mg, and Fe concentration results for fly ash in 0.1 N H_2SO_4 from series R2 are presented in Figure 28. K, Cu, Pb, and Zn concentration results are shown in Figure 29. Representative Zn concentration results using procedure R in series R1 from paint waste using synthetic leachate are presented in Figures 30 and 31. Figure 30 provides more accurate representation of the results of the first few elutions at the various solid-liquid ratios tested, whereas Figure 31 shows the data for more elutions at the 1:10 solid-liquid ratio. The remaining results are in the Appendix.

The results from series R2 indicate that several parameters (Na,K) reached their highest concentrations at the highest solid-liquid ratio. Other parameters (Mg, Fe, Cu, Pb, Zn) display more complex leaching patterns and seem to be influenced by pH. The effect of fly ash solid-liquid ratio on pH is shown in Figure 28. Fly ash neutralizes the acid leachate, and increasing solid-liquid ratios of fly ash increases the pH of the leachate. For a waste which influences the leachate pH, an ideal solid-liquid ratio in a waste replacement procedure would be such that the waste affects the leachate pH during the second or third elution and comes to control pH by the end of the test. More information about the leaching characteristics of such a waste is gained if this occurs. For fly ash in 0.1 N H_2SO_4, 1:10 or 1:20 are good solid-liquid ratios.

Figures 30 and 31 also indicate that higher solid-liquid ratios result in higher concentrations of Zn, at least for this system, and that this effect continues for several elutions. The low concentrations achieved after 3 to 4 elutions were maintained for many additional elutions for this waste-leaching media system, and continuing single elutions over a longer period (over weekends in this case) had little effect on concentrations obtained. Additional data and discussion on elution time in initial elutions will be found in the section titled Influence of Time per Elution.

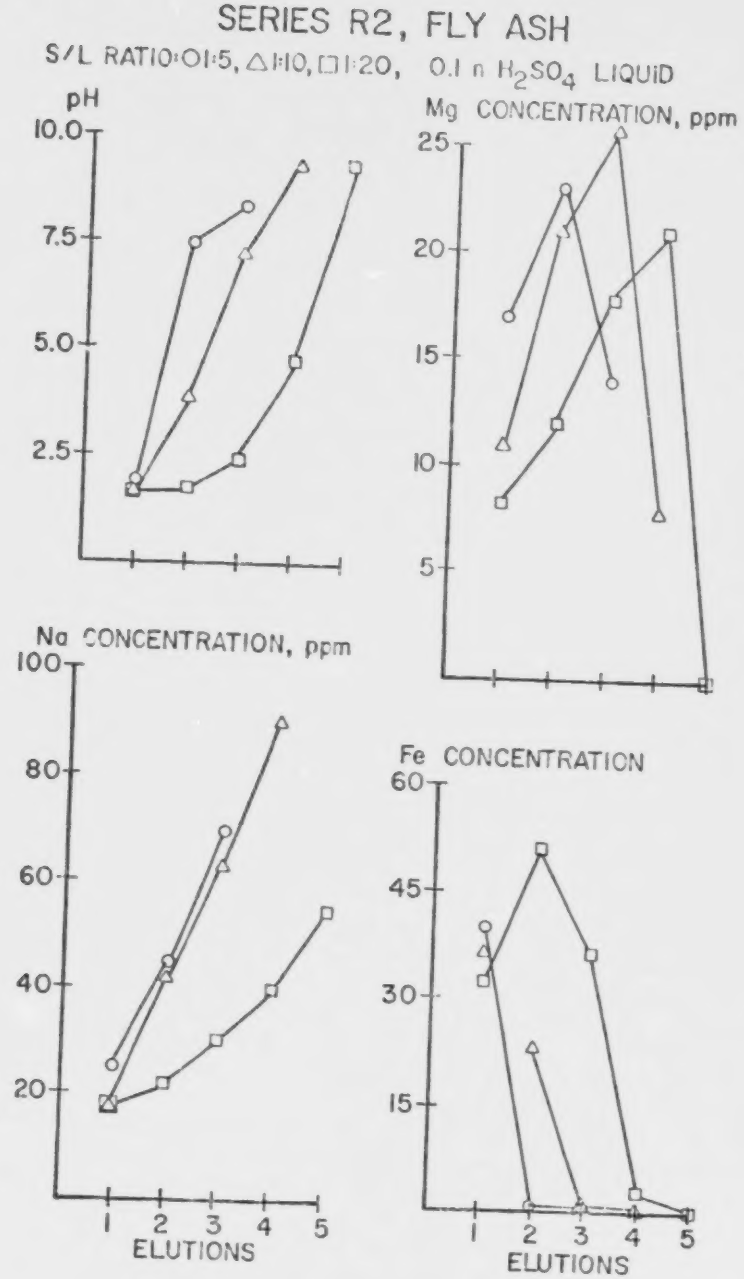

Figure 28. pH, Mg, Na, Fe concentrations for different elutions when fly ash is leached with 0.1N H_2SO_4, series R2.

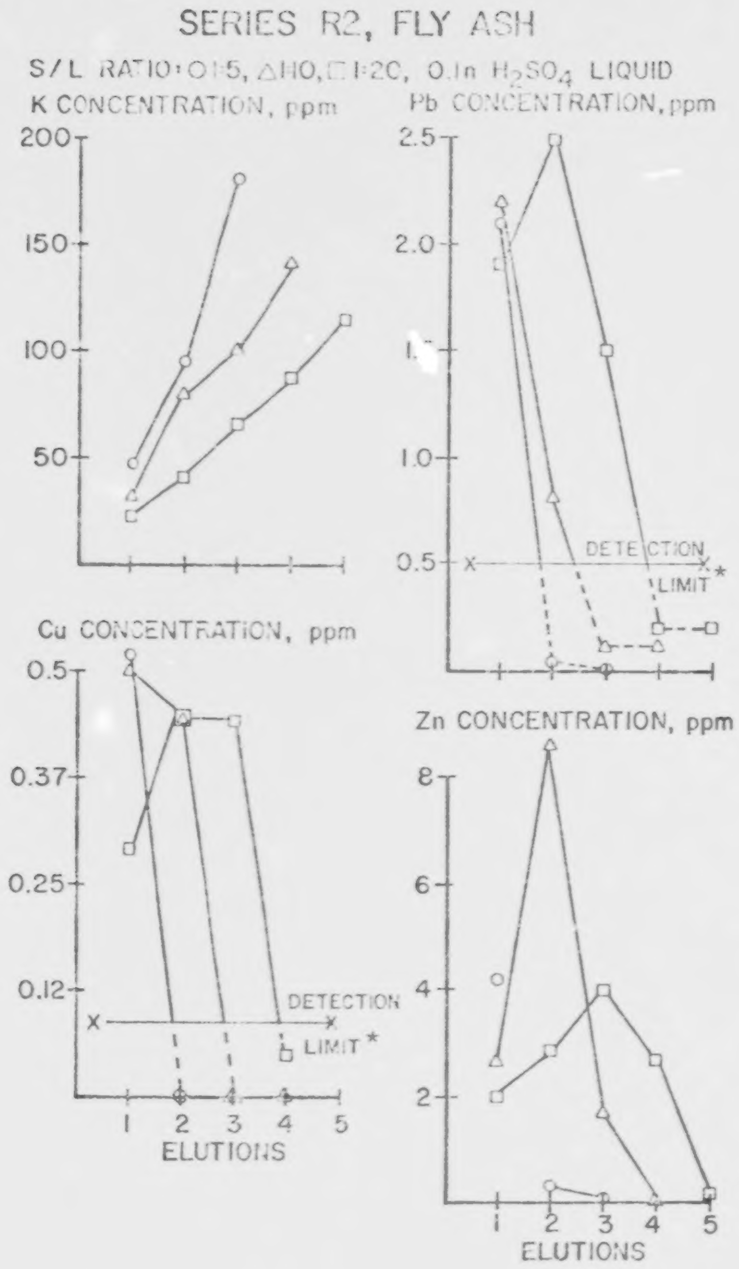

Figure 29. K, Pb, Cu and Zn concentrations for different elutions when fly ash is leached with 0.1N H_2SO_4, series R2.

*All points below detection limits are approximate.

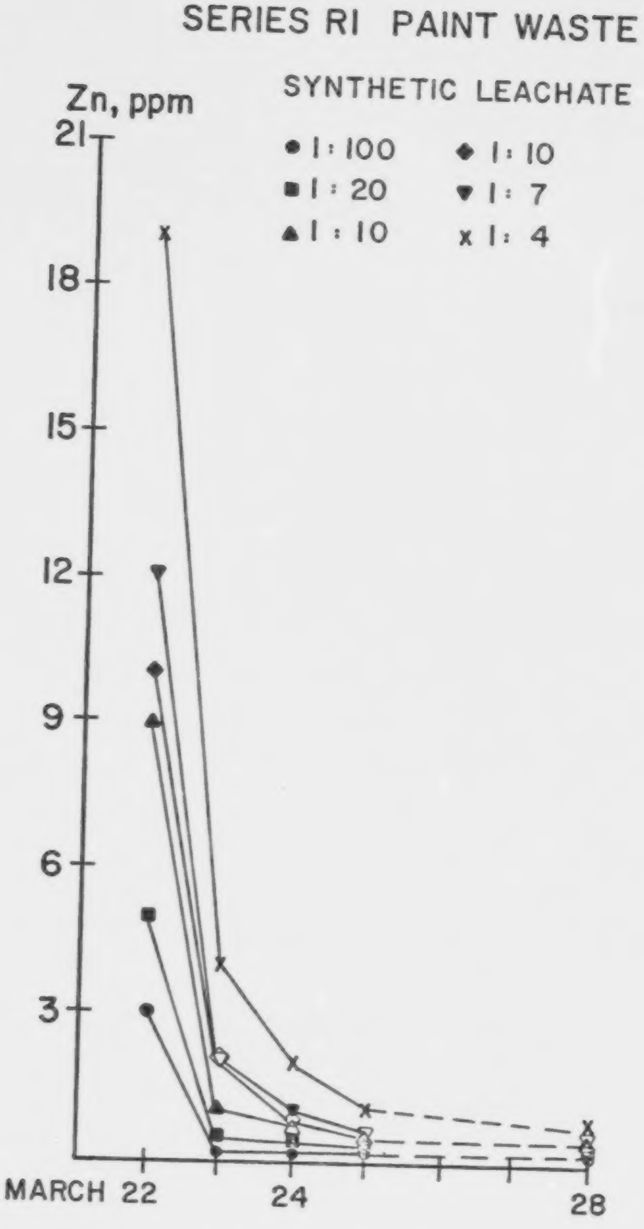

Figure 30. Zinc concentration from paint waste leached with synthetic leachate in series R1 for different elutions and at different solid-liquid ratios.

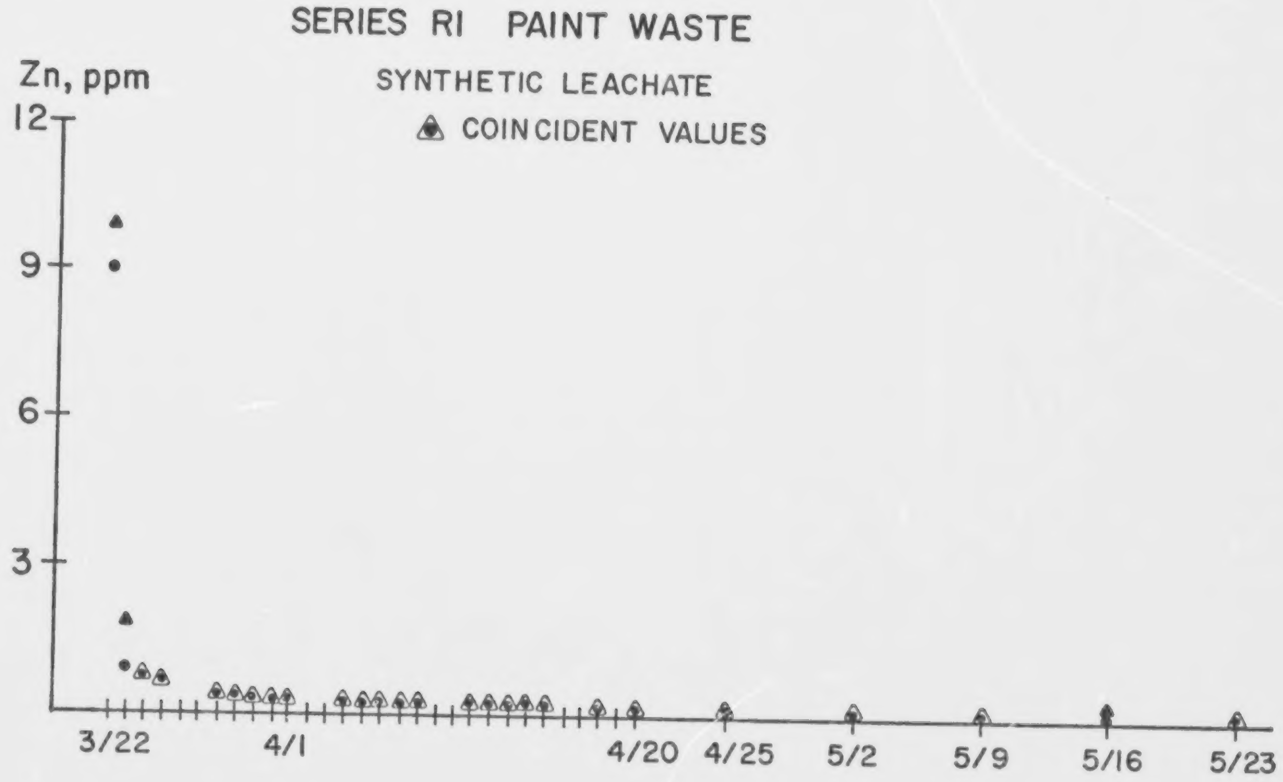

Figure 31. Zn concentration from paint waste leached with synthetic leachate in Series R1 for more elutions at a 1:10 solid-liquid ratio (duplicate runs).

Figure 32 shows the effect of solid liquid ratio on the Na and K concentrations in the third elution leachates from fly ash and paint waste using 0.1 N H_2SO_4 and synthetic leachate. The dotted lines indicate the concentrations expected if the concentrations were directly dependent on solid-liquid ratio, using the concentration at a 1:10 solid-liquid ratio as a basis for calculation. In most cases the experimental results at 1:5 and 1:20 solid-liquid ratios are close to those calculated assuming a direct concentration dependence on solid-liquid ratio, indicating that the concentrations are dependent on solid-liquid ratio.

The results from all of the test series described in this chapter show very clearly the value of using two different test procedures, such as procedures 1 and 5 (C and R) from series V. The importance of keeping the solid-liquid ratio constant and relating it on a dry weight basis for consistency is also shown. It is suggested that a solid-liquid ratio of 1:10 (based on dry weight) be used for Procedures C and R (replacing the waste and replacing the liquid, respectively) in the standard leaching test. The quantity of waste that has to be replaced in Procedure C should be enough to provide the ratios shown in Table 18. Of course, the total quantity of leaching media and waste (ratio 1:10 initially for both procedures) to be used will depend primarily on analytical sample size requirements, but waste availability, and physical and equipment limitations in the laboratory may also be important.

TABLE 18. SOLID-LIQUID RATIOS IN SUBSEQUENT ELUTIONS FOR PROCEDURE C

Elution	
1	1:10†
2	1:7.5
3	1:5
4*	1:2.5

*If required.

†Based on dry weight.

Agitation Methods

In order to obtain uniform solid-liquid contact and representative and reproducible results, a thorough but nondestructive agitation method is needed. Ideally, the method would keep the waste suspended, yet not cause abrasion of the waste particles.

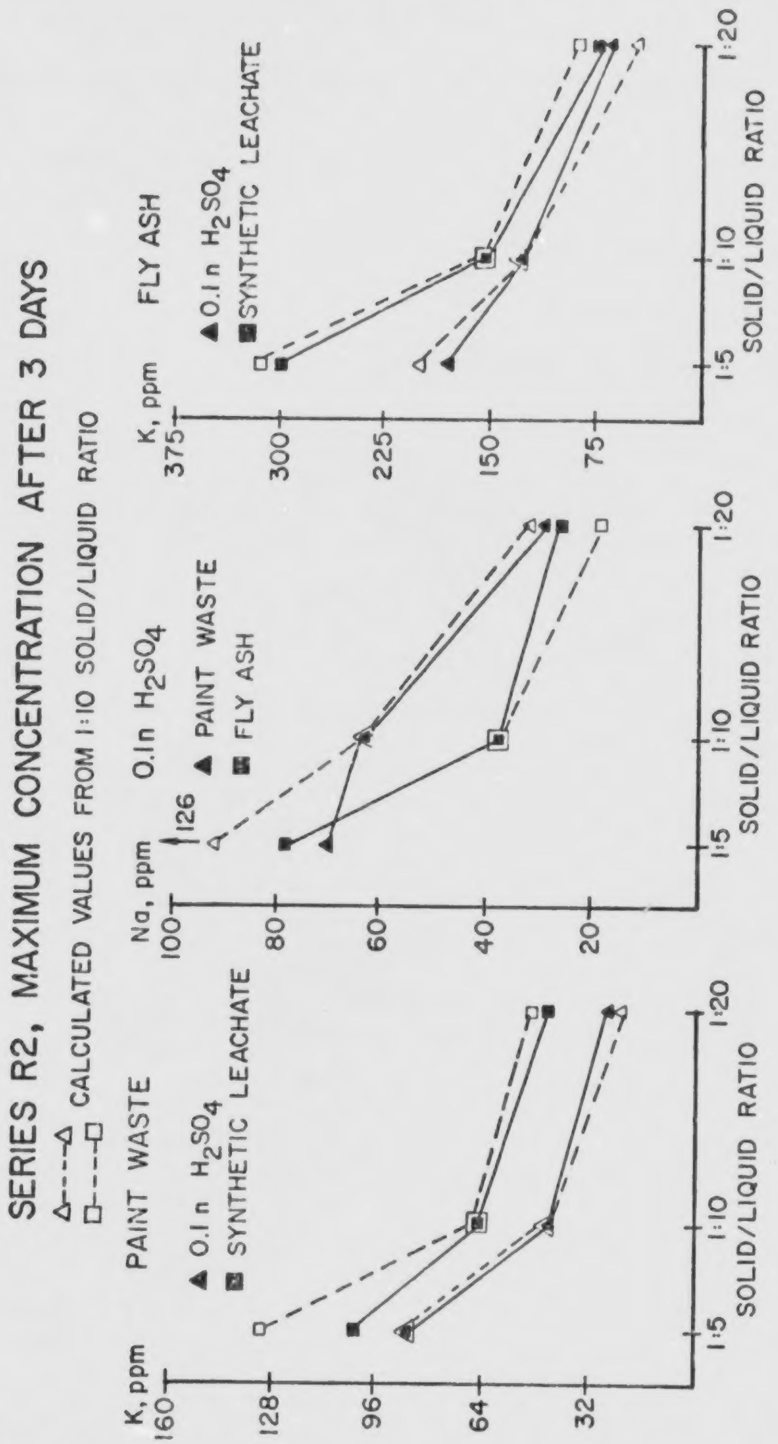

Figure 32. Series R2, maximum concentration after three days for paint waste and fly ash at various solid-liquid ratios.

The influences of different agitation methods were tested in series PV. Five agitation methods were used, selected on the basis of ease of use, availability, and different agitation motions.

1. Continuous shaking (Gyrotory Shaker, New Brunswick Scientific Co.).
2. Continuous mechanical paddle stirring (Phipps and Bird, Inc., Richmond, VA).
3. Intermittent shaking by hand.
4. Swing type shaking.
5. Rotating at two different angles.

Methods 4 and 5 will be discussed in detail later. For the first group of tests (PV1-PV3), agitation methods 1, 2 and 3 were used. The test procedure was that of Procedure R (replacing leaching media). For each elution, 28.6 gm of waste were leached with 200 ml distilled water for 24 hours. Intermittent shaking involves a brief, thorough shaking of a 300 ml Erlmeyer flask by hand three times per day (morning, noon, and late afternoon). Continuous shaking also involved use of a 300 ml Erlmeyer flask, while continuous stirring used a 1000 ml beaker. In both cases speeds sufficient to suspend most of the waste were used.

Typical results using papermill sludge (N) and municipal refuse are presented in Figures 33 and 34. None of the first three agitation methods gave consistently higher leaching rates than the other methods. Stirring gave the highest Ca and Mg release from the papermill sludge and the lowest Ca and Mg release from the municipal waste. With continuous mechanical shaking, the Ca and Mg releases were the highest from the municipal waste and lowest from the paper mill sludge.

Visual observations of the first three mixing methods tested in series PV suggested that none of the three methods provided an optimal solid-liquid contact. In the continuous shaker, the solids often remained at the bottom of the flasks, particularly if a slow shake speed was chosen. Higher speeds seemed to cause changes in the physical properties of some wastes probably due to abrasion. Continuous mechanical stirring may also have caused abrasion for certain of the wastes (i.e., papermill sludge and shredded refuse). Abrasion problems are expected to be of even more concern with grannular materials, as has been experienced with foundry wastes (3). It was also observed that the waste and the liquid tended to move at the same speed as the stirrer in the continuously stirred flask, without optimal mixing.

These observations and the test results caused an examination of other mixing procedures. A swing type shaker that moves slowly over a small angle (approximately 180° every 30 seconds) was test in PV4 and 5. A schematic of this unit is shown in Figure 35. Although this mixer resembles the continuous shaking in the Gyrotory Shaker, the basic idea of a swing type agitation method was found to be worthy of further testing. A machine was designed and constructed specifically for this purpose.

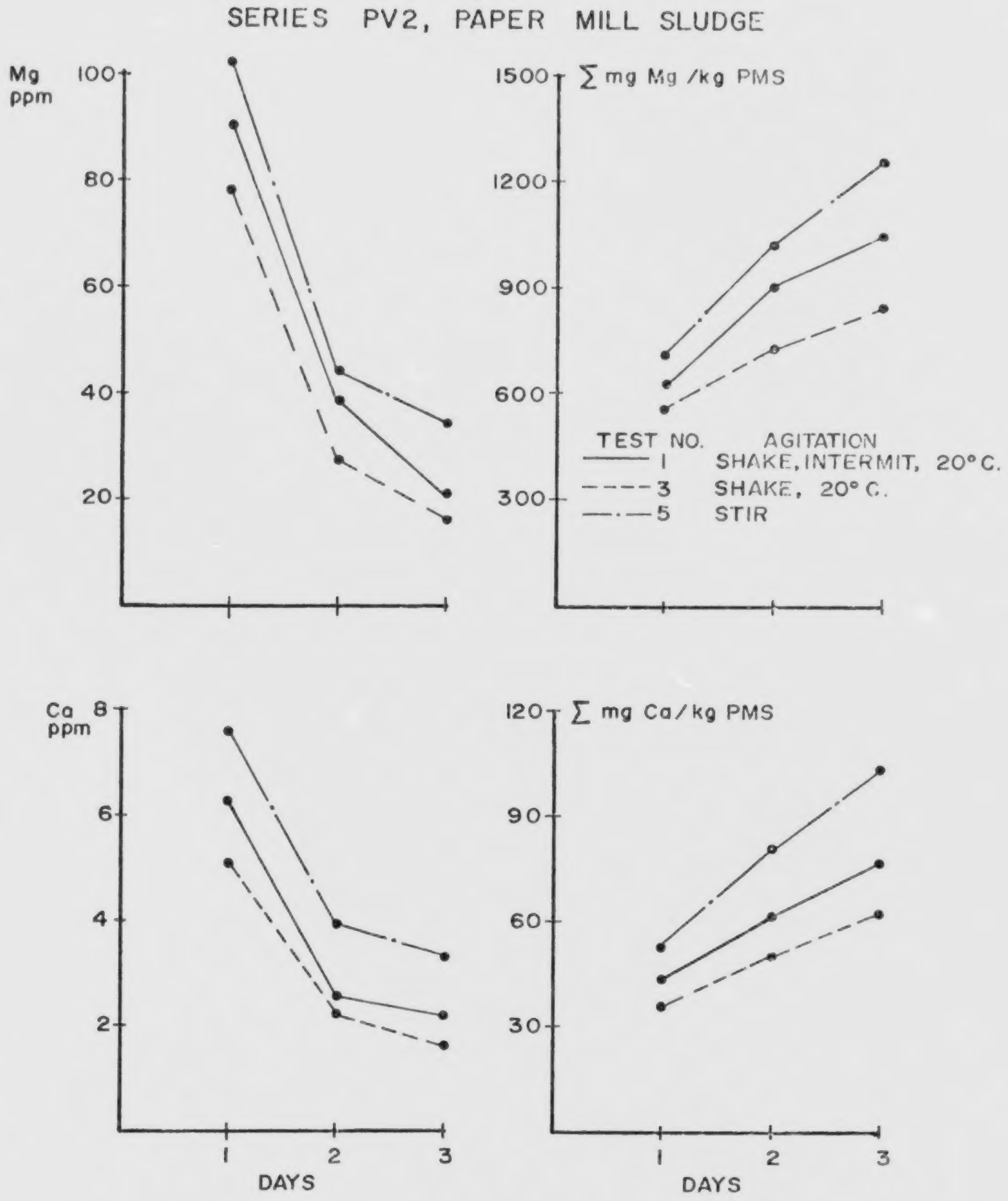

Figure 33. The Ca and Mg results from series PV using papermill sludge (N) and different agitation techniques.

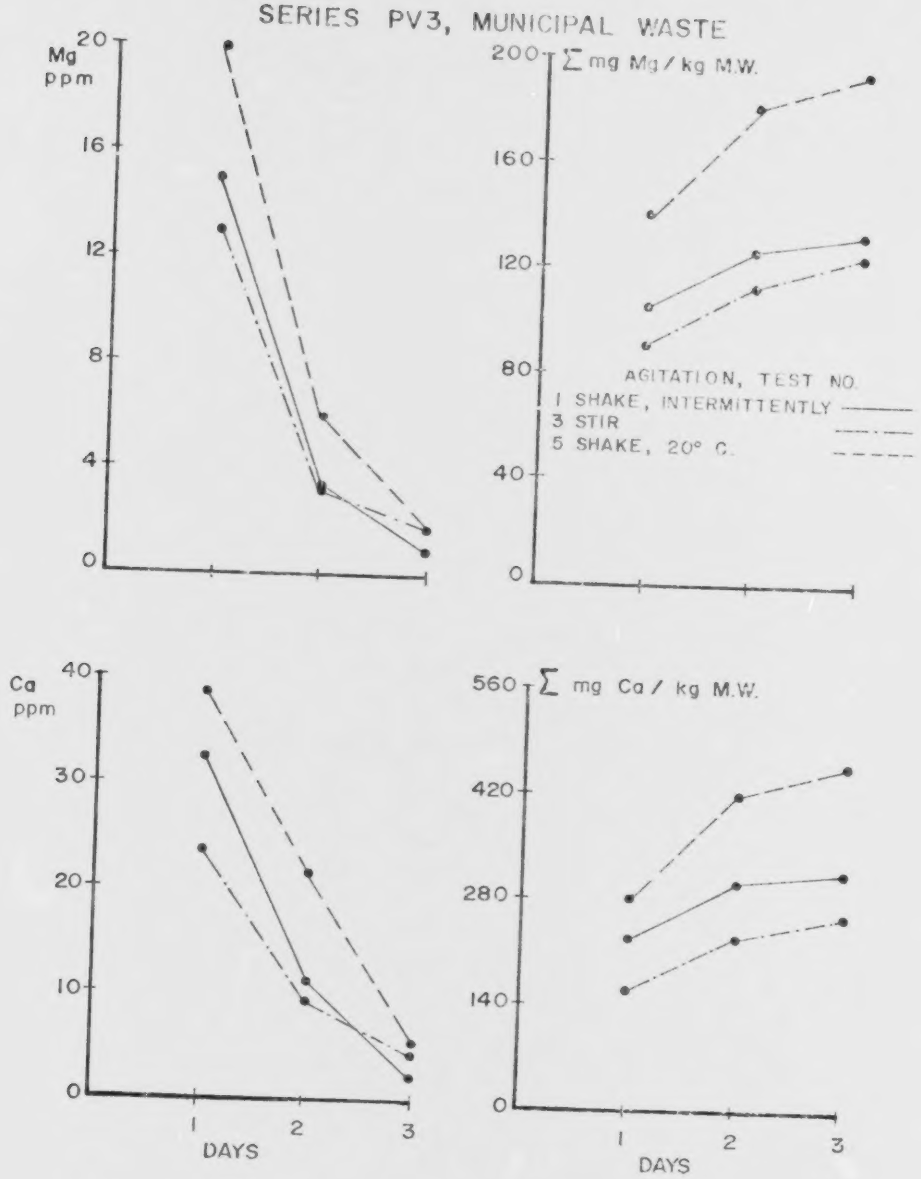

Figure 34. The Ca and Mg results from series PV using municipal refuse and different agitation techniques.

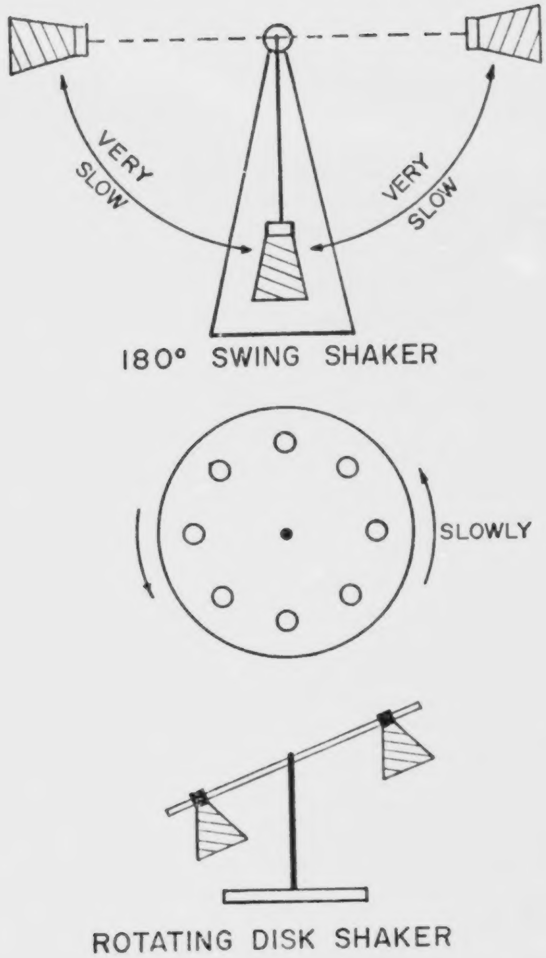
VERY SLOW
VERY SLOW
180° SWING SHAKER
SLOWLY
ROTATING DISK SHAKER

Figure 35. Diagram of the swing shaker and the rotating disc device used in series PV.

Another type of agitation method was also tested in series PV4. This consisted of a rotating disk, which could be tilted at an angle of 20° to 30° from the horizontal (Figure 35). Flasks holding the samples and leaching media were mounted around the periphery of the disk.

In series PV4, a solid-liquid ratio of 1:10 based on dry weight was used. Paper mill sludge (N), clarifier sludge (AA) from an automobile assembly plant, paint waste (AA), and fly ash (Ø < 0.5 mm, AA) were the wastes tested, and distilled water was used as the leaching media. No major differences could be found in the results from the three different agitation methods used. The results are presented in the Appendix. The data were analyzed statistically, but no significant differences between the test results with the three different agitation methods could be found. The statistical tests were made at the 95% confidence level. For example, the mean deviation was 5.75% comparing intermittent with the swing-shaken and 1.41% comparing the intermittent shaking to the rotated samples for the cumulative release of all measured parameters. By observing the mixing in the flasks moving on the rotating disk, it was noted that no visible distinct mixing took place under these conditions. The waste remained in clumps at the bottoms of the flasks. Mixing in the swing-type shaker did not seem to provide good mixing either. The solid often remained on the bottom and on the side walls of the flask without real mixing.

It was possible to tilt the rotating disk such that the angle against the horizontal was 90°. Watching the mixing process of different wastes under these conditions, one noticed a good contact between solids and the liquid especially when square bottles were employed. In round flasks, the waste occasionally slid on the sidewalls and did not mix well. In square bottles this effect did not take place. The rotating speed was in the range of 2-5 rpm.

Vertical rotating disk mixing was tested together with intermittent and swing-shaking (Figure 35) in test series PV5. The same wastes used in series PV4 were used along with shredded refuse from the city of Madison. The procedure was identical with that of series PV4. Square plastic bottles were used in the rotating machine. The liquid in the flasks was 0.1 N H_2SO_4 in order to have a very aggressive leachate, so that eventual differences in the test results could be more readily detected.

Another device tested in this test series was a pressure release valve. During the leaching process, gas production may occur. In order to release the pressure which could damage the flasks or cause leakage, pressure release valves were installed which opened when the pressure in the flask exceeded 1 psi. These were essentially spring loaded ball release valves, with adjustable spring compression. As long as the solid-liquid mixture did not contact the valve, the valve would function properly. If contact was made, the liquid could interfere with the operation of the valve. With the wastes tested, of which the municipal refuse could be considered prone to biological activity, gas production sufficient to warrant the release valve was not observed. Even if a small amount of gas production is anticipated with a particularly biologically active waste, pressure relief valves or frequent manual venting should be used.

The results from series PV5 can also be found in the Appendix. A statistical analysis of the results comparing intermittent and swing-shaking indicates that the cumulative release from all wastes and all parameters measured was 10.9% higher using the swing-shaking procedure. This is at the 95% confidence level. If the agitation procedures of intermittent shaking and rotating are compared, the cumulative release after three days was significantly higher using the rotator (23% more release).

The results from the series PV show that the different agitation methods provide nearly equal release, but that the rotator seemed to be the most effective agitation method, both from visual observations with different wastes, and from somewhat higher release figures. Equipment capable of doing this is commercially available* for a reasonable price, and different rotator heads are available so that different kinds of flasks may be used. The motor did not overheat even after many days of operation. The rotating speed can be varied, but a speed of 3.5 rpm was used for this study. The rotator head should be balanced when performing these tests. The angle of incline of the disk or head is adjustable; the angle of the circle of rotation to the horizontal was set at approximately 70° for this research. Visual observations should be used to set the angle and speed so wastes are mixed and turned to provide good media-waste contact without causing any more abrasion than necessary.

Influence of Time per Elution (Series P1)

Fly ash with a particle size smaller than 0.5 mm and paint waste (AA) were used in series P1 to investigate the influence of reaction time on the rest results. Three elutions of various durations were used. After each elution, the liquid was filtered for analysis, and the waste was contacted with fresh leachate (Procedure R). Three different leachates were used: 0.1 N H_2SO_4, synthetic leachate, and distilled water. The following reaction times were used for each elution: 2 hours, 24 hours, 48 hours, and 72 hours.

The cumulative release after 3 elutions for each parameter is plotted in Figures 36 and 37. There is no consistent trend in cumulative release after 3 elutions with respect to reaction time. In particular, there was an increase in cumulative release with reaction time for the test with fly ash in 0.1 N H_2SO_4. This tendency is obvious for K, Mg, and Fe. In this case, it is possible that parts of the fly ash are dissolved more readily by the acid with increasing reaction time, which suggests that the fly ash may change its composition and pore size during

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*Such as the Rugged Rotator, model RD-250, manufactured by Kraft Apparatus, Inc., of Mineola, New York, as used in this study.

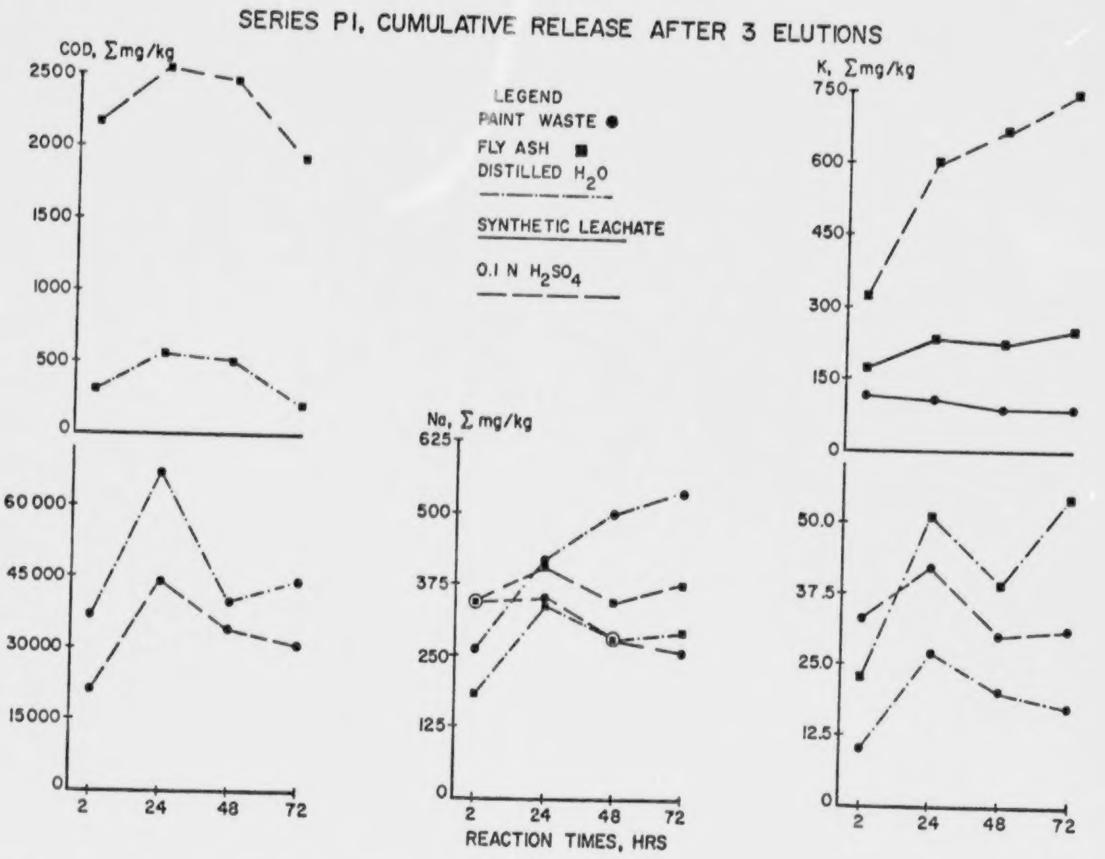

Figure 36. Cumulative release after three elutions for series P1 (COD, Na, K) for different reaction times.

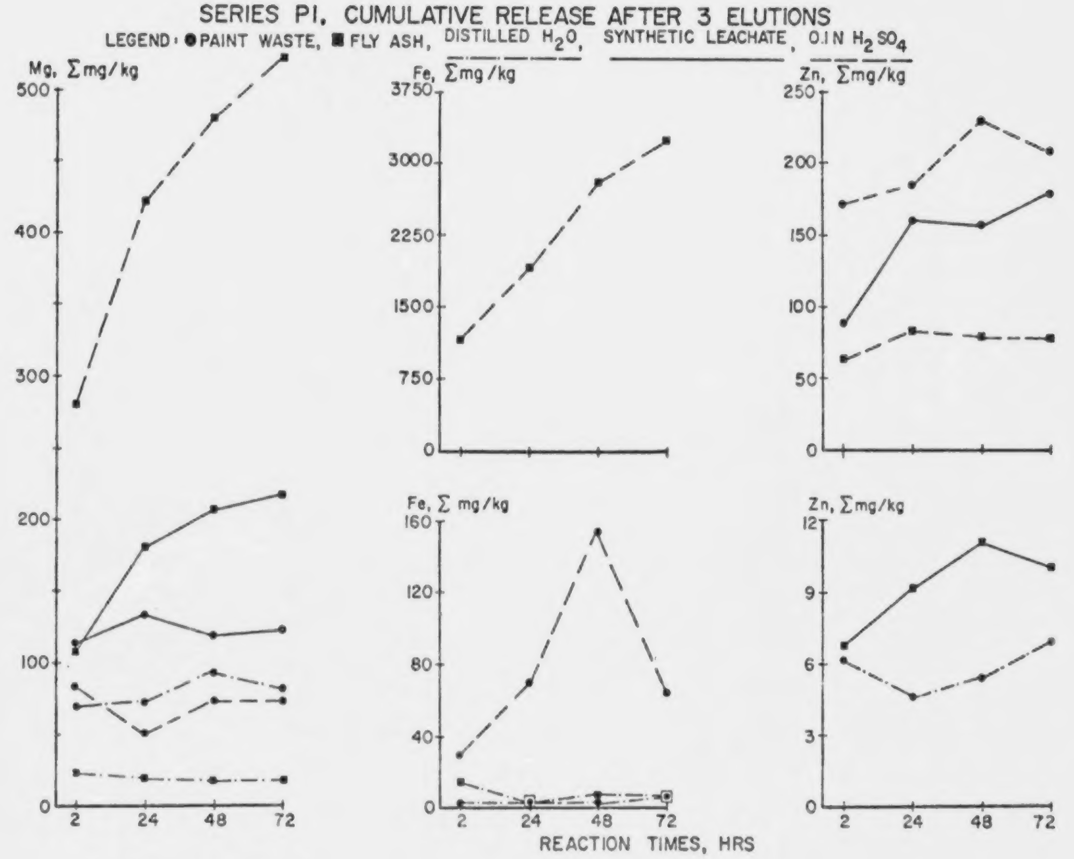

Figure 37. Cumulative release after three elutions for series P1 (Mg, Fe, Zn) for different reaction times.

the test. This does not happen to the same degree when distilled water and synthetic leachate are used. With the paint waste, the cumulative release is more independent of elution time.

In practically all cases with both wastes, the cumulative release is lower with a two-hour reaction time, suggesting the use of a longer time. For reaction times 24 hours or greater, the cumulative release is varied—sometimes rising, sometimes falling—indicating that the effect of reaction times greater than 24 hours are not consistent. Therefore, selection of a leaching time within this 24-72 hour range should be made on factors other than approach to equilibrium conditions, since these systems do not appear to be in equilibrium within this time frame.

A 24-hour elution time is normally convenient for laboratory organization and personnel. Since 3 elutions are required by this test, an elution time of 48 hours would make the total test 144 hours long and would necessitate weekend laboratory activity. Also 48-hour or 72-hour elutions would introduce greater possibility of biological effects in the test system. If higher leaching rates are desired, it is more effective to use more elutions of 24 hours reaction time each than to have a longer reaction time with fewer elutions. (Note that the number of elutions and the total test time will be discussed later.)

Influence of Number of Elutions

Ideally, a test should run until no further material leaches out of the waste. This is not practical because small amounts of material may leach from a waste for years.

The number of elutions required to establish release patterns is a function of the test procedure (i.e., Procedures R or C), the solid-liquid ratio, the kind of eluent used, agitation method, the length of each elution, etc. From previous sections, information is available to set reasonable values for these test variables so the influence of the number of elutions can now be evaluated.

Test series R1 was run to investigate the influence of the solid-liquid ratio and the number of elutions on the results of the leaching test. Portions of these results were cited previously in the section on solid-liquid ratio. In this test series, Procedure C was used for fly ash (AA) and paint waste (AA). This procedure uses a solid-liquid ratio of 1:10 on a dry weight basis, and the waste is replaced for successive elutions. A total of 31 elutions over a 14-week period were made. The first 21 elutions, done in 4 weeks, used an elution time of 24 hours, except on weekends when a 72-hour elution was performed. For elutions after the 21st, the elution time was extended to one week. Longer elution times were used to reduce the number of samples and yet retain a long test duration. The results from test series R are presented in the Appendix.

Two examples of the results are shown in Figures 38 through 41 to illustrate the results that were obtained. Figures 38 and 40 provide results for the solid-liquid ratios tested for the first few elutions. Figures 39 and 41 show the results at a solid-liquid ratio of 1:10 over more elutions. Figures 38 and 39 indicate that the release of Fe from fly ash by using 0.1 N H_2SO_4 takes a long time. The release pattern suggests that fly ash is not a uniform waste. The increase of the Fe concentrations after 3 elutions is difficult to explain, because this did not occur when solid-liquid ratios other than 1:10 were used. On the other hand, the test at a 1:10 ratio was performed in duplicate and the results were the same in both. In Figures 40 and 41, the results for Zn indicate that for paint waste and 0.1 N H_2SO_4, a steady state is reached after three or four elutions. For the fly ash, a more complex release pattern is indicated.

An evaluation of the percentage of the total release found by each elution was made in the following manner. First, the number of elutions needed to reach steady state conditions was determined. Steady state conditions were assumed to exist when the concentration stopped falling and maintained a relatively stable level (see Figure 42). The total release up to the time when steady state conditions were reached was calculated, and the percentage of the total released by each elution determined. In cases where steady state conditions were not reached, total release at the end of the test was used at 100%. The percentage released by each elution for series R1 is shown in Figures 43 and 44.

As seen in Figures 43 and 44, some parameters (e.g., Zn in paint waste) approach 100% release within a few elutions, while others (e.g., Fe in fly ash) release very slowly. Therefore, the ideal number of elutions varies with the parameter observed. In two cases (Fe and K from fly ash in 0.1 N H_2SO_4), steady state (basic) conditions were not reached by the end of the test. Percentage cumulative release curves are either straight lines or have decreasing slope with increasing elution numbers. The fastest release, which corresponds to the steepest slope, occurs in the first test elutions.

In Table 19 the results from test series R1 are summarized. Cumulative release from elutions before steady state values are reached is compared with the total release after 28 elutions (11 weeks). The results indicate that before the steady state values are reached, the cumulative release as a percentage of the total release over the whole test period, varies between 40 and 100%.

The total release from series R1 after 11 weeks (28 elutions) was also compared with the results from total digestion of the fly ash. Although it was not possible to dissolve completely all solids (residue ~1-5%), the total release after 28 elutions was relatively low compared with the amount of each component in the original fly ash. The results are presented in Table 20. The total release from the fly ash was higher when 0.1 N H_2SO_4 was used in the leaching test.

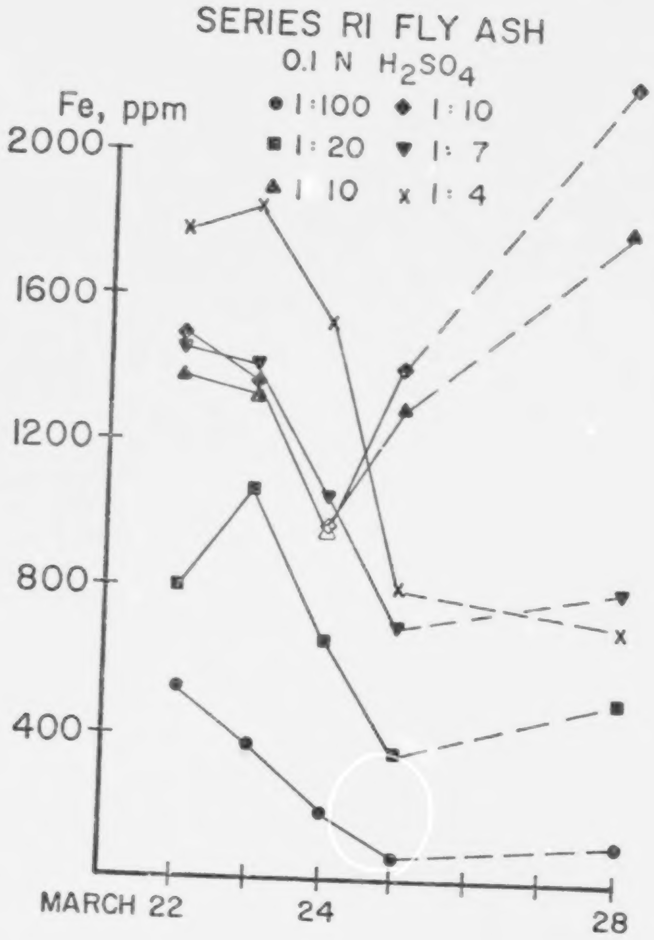

Figure 38. Fe concentrations from long-term leaching experiments in series R1 with fly ash and 0.1N H_2SO_4 at various solid-liquid ratios.

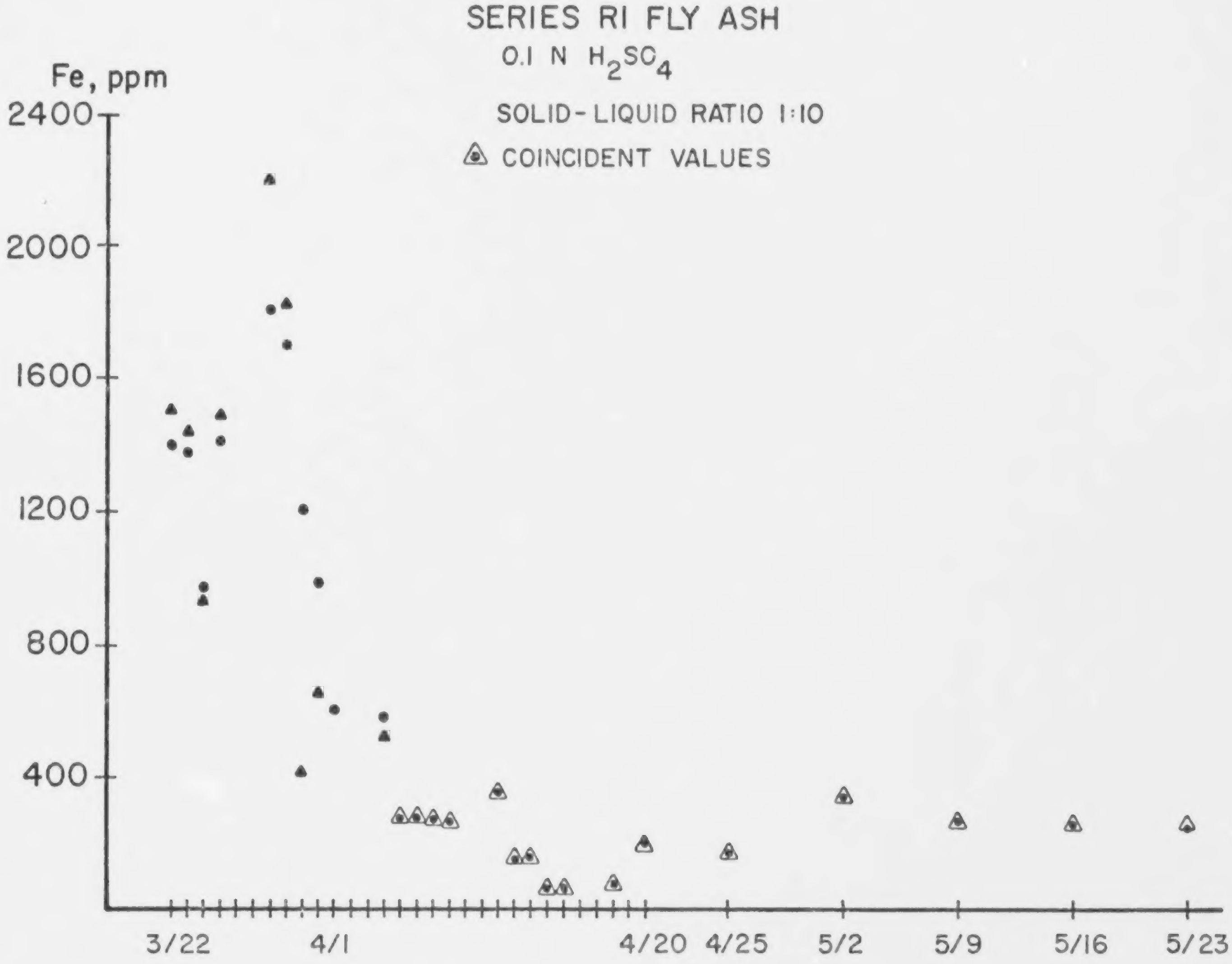

Figure 39. Fe concentrations from long-term leaching experiments in series R1 with fly ash at solid-liquid ratio of 1:10 (long-term results, duplicate runs).

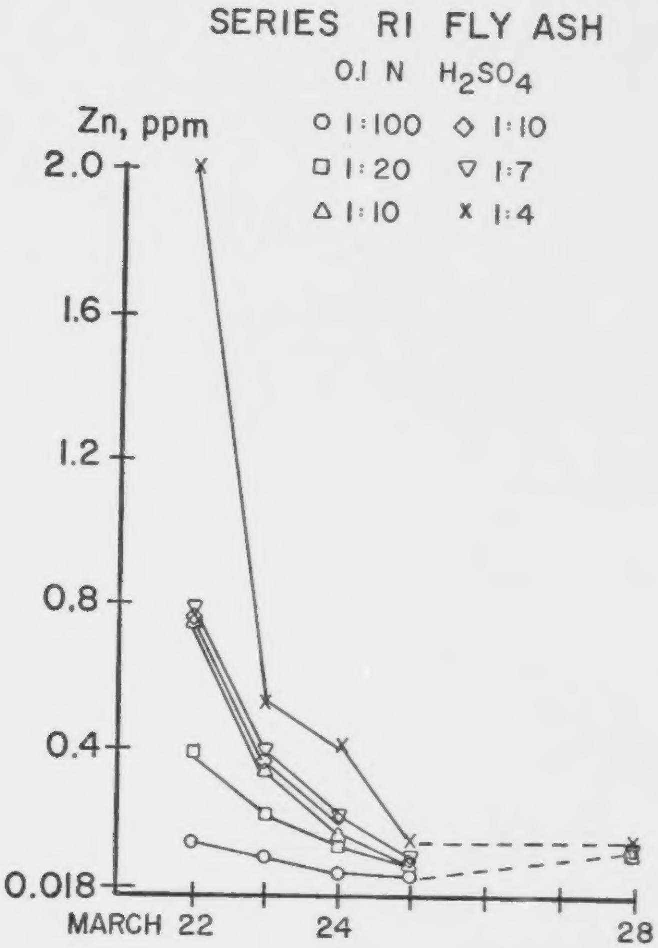

Figure 40. Zn concentrations from long-term leaching experiments in series R1 with fly ash and 0.1N H_2SO_4 at various solid-liquid ratios.

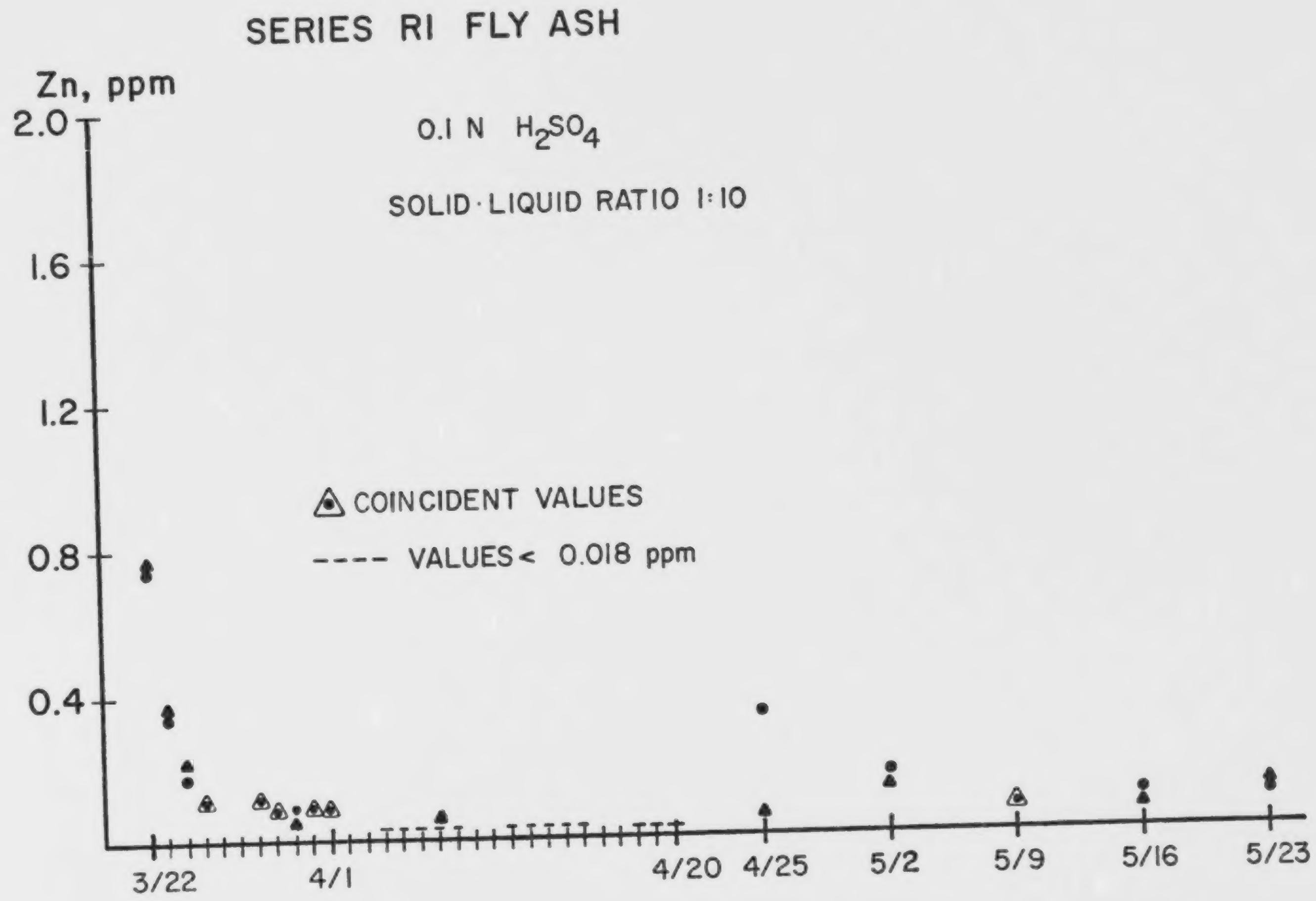

Figure 41. Zn concentrations from long-term leaching experiments in series R1 with fly ash at solid-liquid ratio of 1:10 (long-term results, duplicate runs).

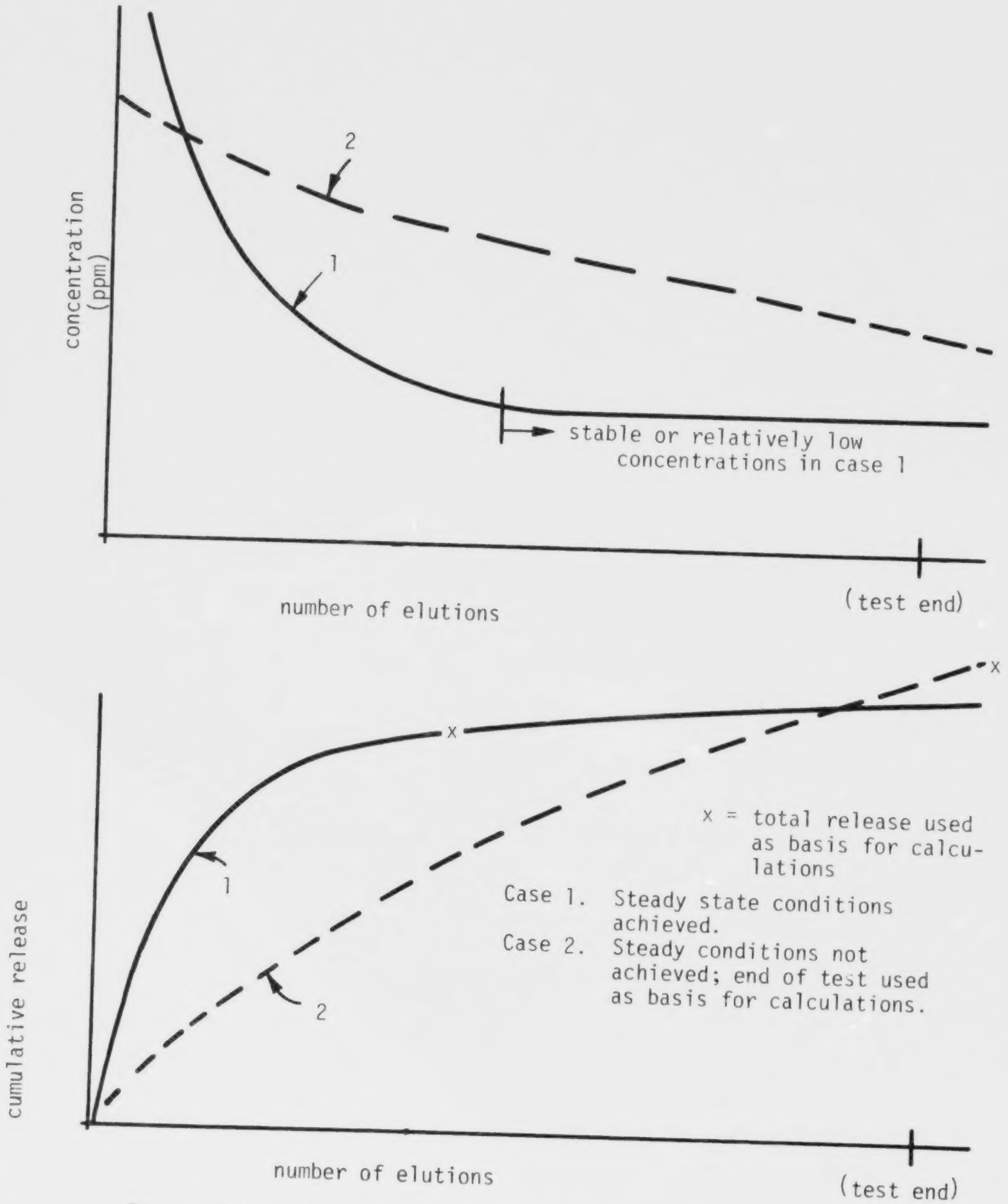

Figure 42. Conceptual basis for calculation of the percentage of cumulative release for each elution in Figures 43 and 44, and Table 19, as indicated by (1) stable concentration levels attained or (2) no stable values attained before the test was terminated.

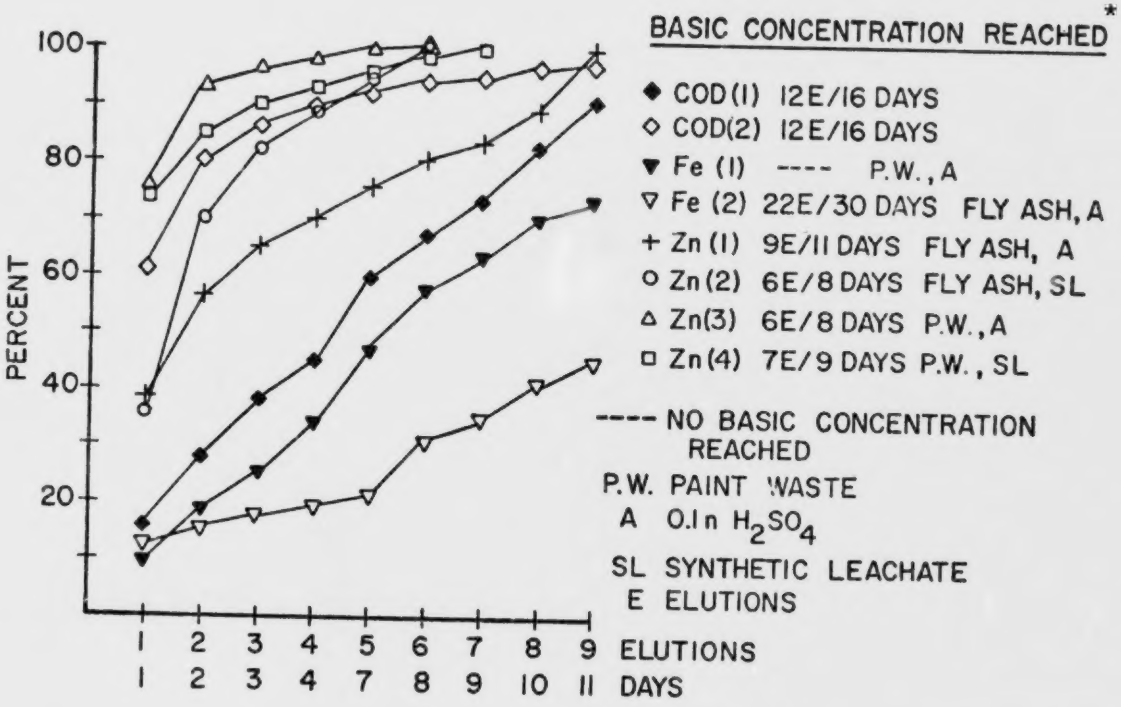

Figure 43. Cumulative release as a percentage of the basic or steady state concentration for various parameters leached from paint waste and fly ash in series R1.

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*Basic concentration equals steady state or end of test concentration, as defined in Figure 42.

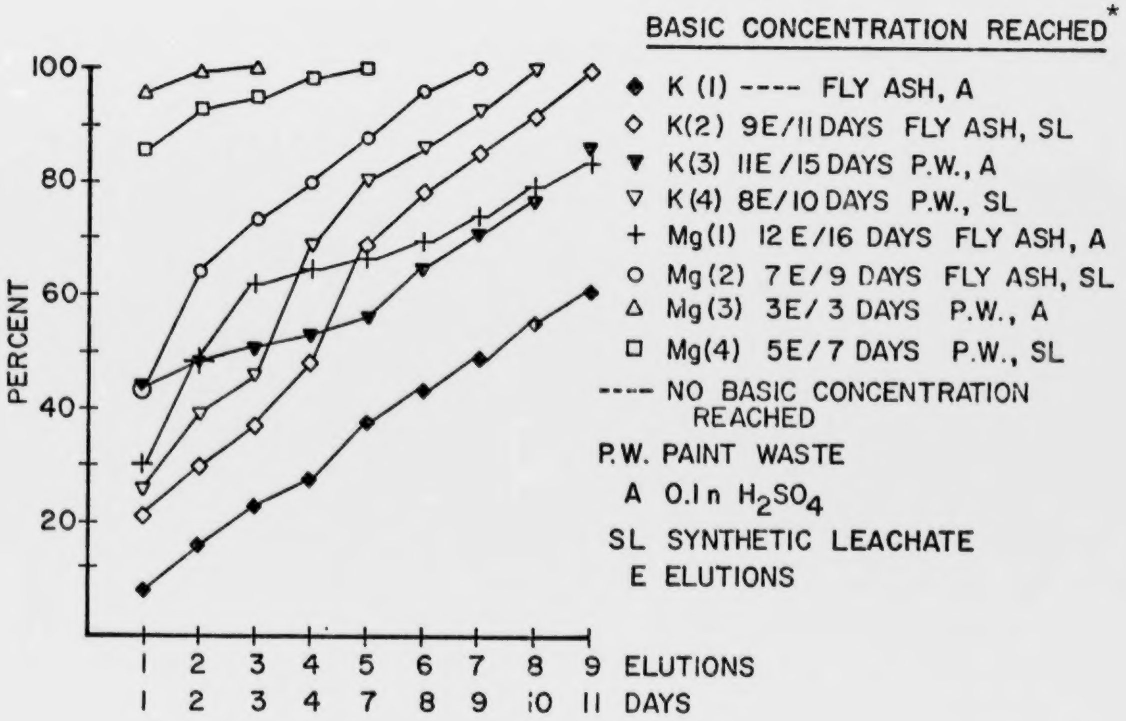

Figure 44. Cumulative release as a percentage of the basic or steady state concentration for various parameters leached from paint waste and fly ash in series R1 (continued).

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*Basic concentration equals steady state or end of test concentration, as defined in Figure 42.

TABLE 19. COMPARISON OF THE CUMULATIVE RELEASE WHEN STABLE LEVELS ARE REACHED TO THE CUMULATIVE RELEASE AFTER A TEST PERIOD OF 11 WEEKS (28 ELUTIONS) IN PERCENT

Parameter	Eluent	Waste	No. of elutions (days) to reach stable levels Elutions	 Days	Release when stable levels attained as a percentage of total cumulative release after 11 weeks (28 elutions)
COD	0.1 N H_2SO_4	Fly Ash	12	16	72.8%
COD	0.1 N H_2SO_4	Paint Waste	12	16	87%
Na	0.1 N H_2SO_4	Fly Ash	not reached		-----
Na	0.1 N H_2SO_4	Paint Waste	9	11	40.5%
Mg	0.1 N H_2SO_4	Fly Ash	12	16	69.8%
Mg	Synth. Leach.	Fly Ash	7	9	51.6%
Mg	0.1 N H_2SO_4	Paint Waste	3	3	100%
Mg	Synth. Leach.	Paint Waste	5	7	78.8%
K	0.1 N H_2SO_4	Fly Ash	not reached		-----
K	Synth. Leach.	Paint Waste	9	11	46.2%
K	0.1 N H_2SO_4	Fly Ash	11	15	63.7%
K	Synth. Leach.	Paint Waste	8	10	46.9%
Fe	0.1 N H_2SO_4	Fly Ash	not reached		-----
Fe	0.1 N H_2SO_4	Paint Waste	8	11	41.8%
Zn	0.1 N H_2SO_4	Fly Ash	9	11	79.2%
Zn	Synth. Leach.	Paint Waste	6	8	70%
Zn	0.1 N H_2SO_4	Fly Ash	6	8	98.6%
Zn	Synth. Leach.	Paint Waste	7	9	83.4%

TABLE 20. CALCULATION OF THE RELEASE AFTER 28 ELUTIONS (11 WEEKS) AS A PERCENTAGE OF THE AMOUNT OBTAINED BY TOTAL DIGESTION OF FLY ASH

Percentage of Dry Weight of Different Parameters in Fly Ash*	Absolute Content in 25 gm Fly Ash (mg)	Cumulative Release [mg/25 gm] Over Test Period (11 weeks)	Percentage Release After 11 Weeks and 28 Elutions
Na = 18.5%	4625	(1) 3.22	0.07%
K = 1.54%	385	(1) 6.59 (2) 1.13	1.71% 0.29%
Pb = 0.12%	30	-----	-----
Fe = 8.66%	2165	(1) 150.11	6.93%
Mg = 0.51%	127.5	(1) 1.63 (2) 0.52	1.28% 0.41%
Cu = 0.03%	7.5	-----	-----
Zn = 0.047%	11.75	(1) 0.025 (2) 0.024	0.21% 0.20%
Total 29.4%			

(1) 0.1 N H_2SO_4

(2) Synthetic leachate.

*Measured after total digestion according to procedure specially developed for fly ash digestion (18).

The results indicate that with some wastes, elution at steady state conditions may continue over a very long time period or that portions of the chemical constituents are not leachable (i.e., bound in the particles and not available for leaching). It is clear, at least for the wastes tested, that total release is not a realistic goal for any reasonable leaching test. A complete digestion procedure, if available, would not relate to landfill practice unless the particular waste in question does in fact dissolve or decompose completely in the landfill. Conversely, more elutions than 3 to 5 were found in previous results to provide little or no additional information unless complete long-term leaching procedures are the alternative.

It is clear that there is no ideal number of elutions for all parameters in all wastes. Three elutions were chosen for the S.L.T. on the basis of convenience and lack of data suggesting otherwise. Tests can be started on Monday, elutions finished by Thursday and analytical work done on Friday. Four elutions could also be used without inconvenience. Steady state concentrations reached after a large number of elutions are typically low and are influenced by the background values of the leachate. Also steady state concentrations are often near parameter detection limits and analytical errors increase.

If the results from 3 elutions indicate continuing release at about the same rates, additional elutions might be performed, but the results must be clearly marked as being from more than 3 elutions and must be interpreted accordingly. Continuing release probably indicates that the waste is dissolving or breaking down, and that eventually those chemical species being leached will be released entirely to the leaching media. In such a case, a complete analysis, such as would be obtained by digestion, for example, might be useful.

Influence of Temperature and Biological Aspects

Temperature will have an effect on leaching test results due to its effect on chemical and biological reactions. Two tests, PV2 and PV3, which were presented earlier in the discussion on agitation methods, also examined temperature effects at 20°C and 33°C. Results beyond those presented in the earlier section can be found in the Appendix. Statistical evaluation of the results indicates that the variation due to the 13°C temperature difference was approximately the same as the variation due to analytical error. In one case (PV2), the deviation due to temperature was 15.1% lower than that due to analytical error, while in the other case (PV3), the deviation due to temperature was 6.7% higher. It is concluded that for the wastes tested, and over the narrow but realistic temperature range tested, the influence of temperature was not important. Neverthless, it is obvious that laboratory temperatures should be used as a standard, and unusual test temperatures be documented and reported. Such might be the case, for example, if the waste in question is being landfilled under unusual temperature conditions.

Two types of tests were conducted to examine biological aspects of leaching an industrial waste.

The first biochemical test was a toxicity test to determine the toxicity of the waste to methane producing bacteria, such as would be found in an actively decomposing sanitary landfill. A diagram of the apparatus used for the toxicity test is shown in Figure 45. The gas volume was measured by the displacement of water in the plastic column shown in Figure 45. The column on the right is used for pressure adjustment.

The industrial waste was mixed with an active culture of methane producing bacteria in a shredded refuse—sewage sludge slurry (Refuse + refuse water to sludge weight ratio of 2:1), incubated at 33°C and the resulting methane volume produced compared with the volume produced from an industrial waste-free mixture. A typical gas production curve from the municipal refuse-sludge mixture is shown in Figure 46, where the gas production indicates a lack of toxic effects for this waste mixture. Should another waste prove toxic to the methane producing bacteria in the test, it would probably also be toxic to the bacterial populations in a sanitary landfill, and might be better disposed in another type of landfill (assuming decomposition is not to be inhibited).

The second biochemical test was a comparison of the leaching characteristics of Madison municipal refuse when leached with distilled water and distilled water with bacteria inhibiting agents added. Unfortunately, most bacteria inhibiting agents directly affect the leaching properties of the eluent. Three inhibiting agents—$AgNO_3$, $CuSO_4$ and thymol—were used in separate tests. Test results that are similar in all inhibited leachates but vary between inhibited and distilled water leachates can be considered to reflect bacterial inhibition. Test results that vary between the three inhibited leachates probably reflect the effects of the inhibiting agents.

Tests B2 and B3 used a solid-liquid ratio of 1:7 based on wet weight and Procedure R (new leaching media for each elution). The moisture content of the refuse was approximately 50%. B2 was run at room temperature for three days, while B3 was run at 33°C for seven days. The bacterial inhibiting agents were added at the following concentrations:

$AgNO_3$	1.0 g/l
Thymol	1.5 g/l
$CuSO_4$	1.5 g/l

In Test B3, it was possible to qualitatively assess anaerobic biological processes by smell. Only the distilled water leachate had the volatile organic acid smell typical of anaerobic processes.

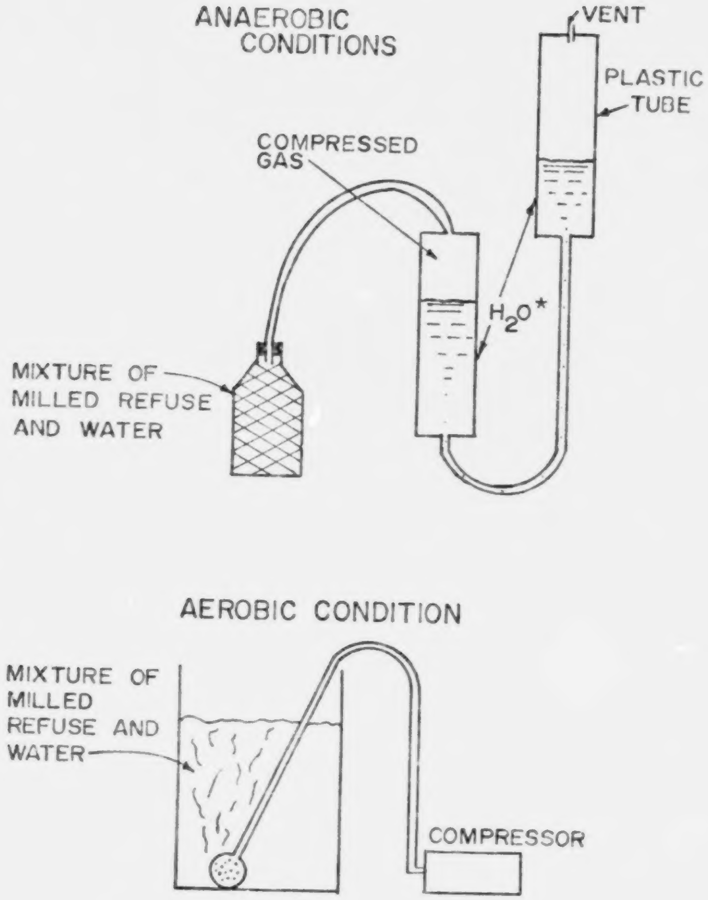

ANAEROBIC CONDITIONS
VENT
PLASTIC TUBE
COMPRESSED GAS
H_2O*
MIXTURE OF MILLED REFUSE AND WATER
AEROBIC CONDITION
MIXTURE OF MILLED REFUSE AND WATER
COMPRESSOR

Figure 45. A diagram of the toxicity test apparatus.

*Equalize water levels during gas volume measurement.

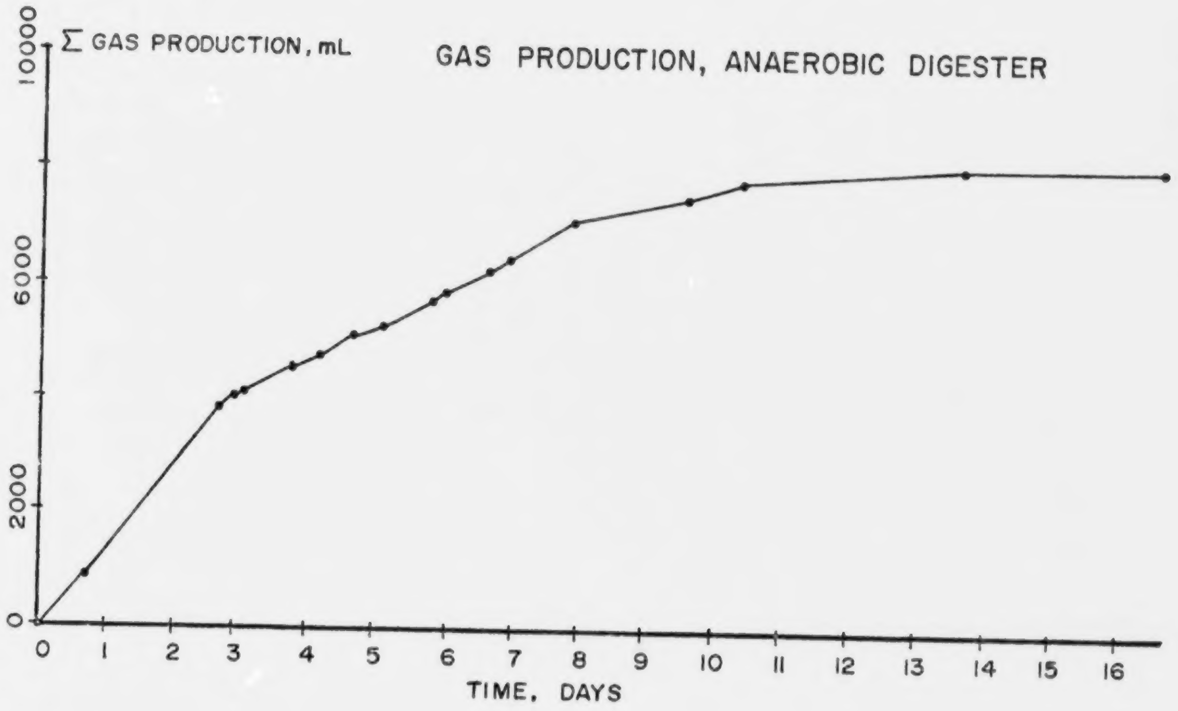


Figure 46. The volume of methane produced versus time by a municipal refuse-sewage sludge slurry.

The last test for determining biological influences was B4. Here Procedure C was used (new waste for each elution)with the shredded refuse being replaced two times. The solid-liquid ratio was 1:10 based on dry weight. The concentrations of the bacterial inhibiting agents were 10 g/l for all agents. At the conclusion of each elution, 200 mls of the filtered leachate was removed for analysis.

A summary of the cumulative release from test B2 and the maximum concentrations in tests B3 and B4 are shown in Figures 47 and 48. Complete results are given in the Appendix. As can be seen from the figures, there is no consistent effect of the bacterial inhibiting agents on the test results. This indicates that any effects of bacteria on the leaching characteristics of the waste are less than the effects of the bacterial inhibiting agents or of random test variations. Apparently within the time span of the leaching tests used, bacterial action did not show an overwhelming effect on leaching from a biologically active waste, and so no special precautions are required in the tests to insure inhibition of bacterial action with wastes that are at least no more active biologically than shredded refuse.

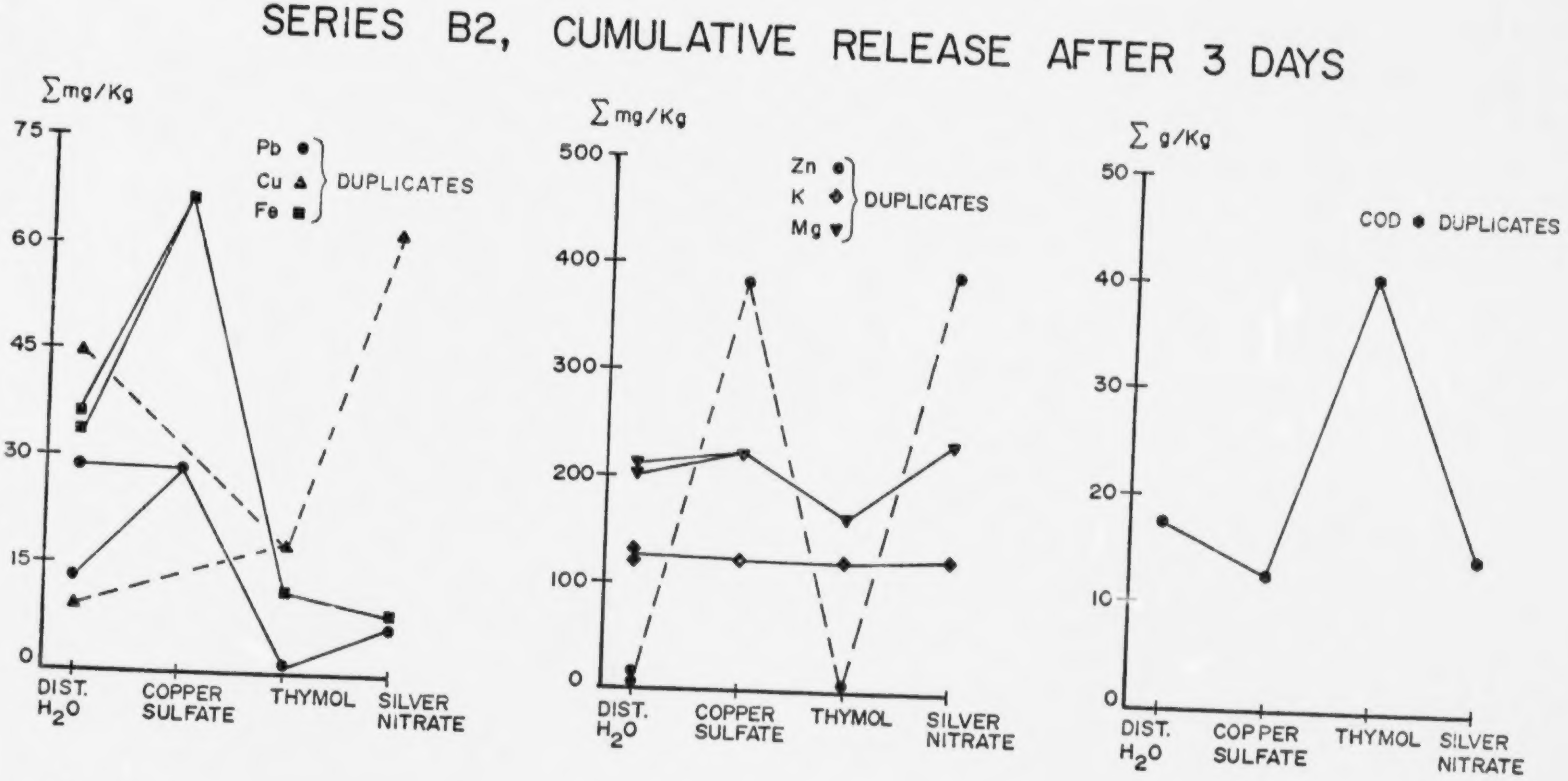


Figure 47. Cumulative releases of several parameters for municipal refuse leached with distilled water containing bacterial inhibiting agents, series B2.

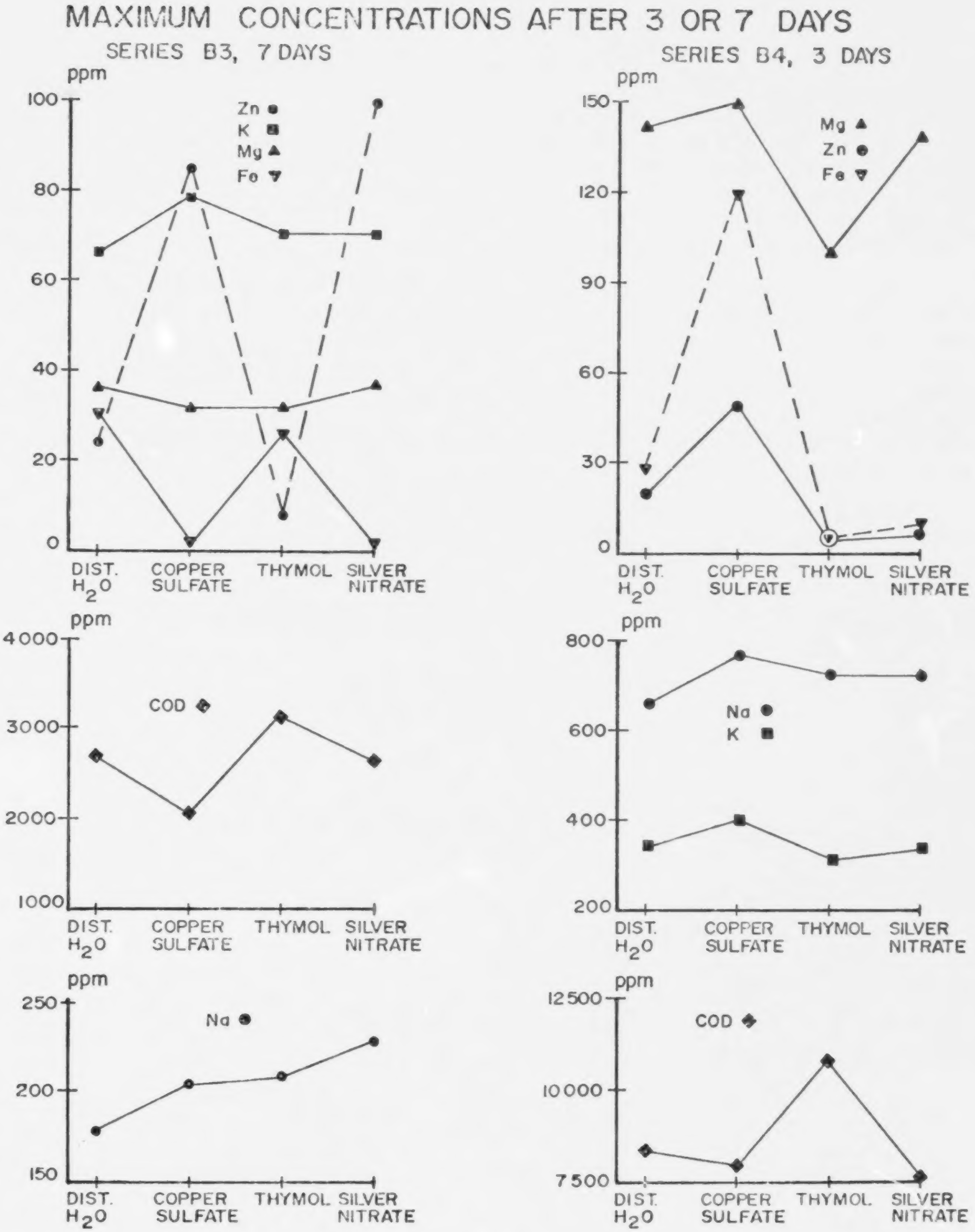


Figure 48. Maximum concentrations of several parameters for municipal refuse leached with distilled water containing bacterial inhibiting agents, series B3 and B4.

SECTION 7

SUGGESTED PROCEDURE FOR A STANDARD LEACHING TEST

This chapter will provide an overview of the suggested leaching test procedure. Figure 49 shows a flow scheme emphasizing the sample preparation steps of the suggested standard leaching test. In general, initial sampling of a waste will be done by non-laboratory personnel. Obviously, major errors can arise from improper sampling, but this aspect of the test was not included in the present study. It must be assumed that the sample is representative of the waste that is to be tested and possibly landfilled.

Unless the waste is homogeneous, subsampling must be done carefully, and innovative techniques will often be necessary. Several different techniques may be required. Some techniques have proven useful within this study. Sand-splitters are good for dividing solid, granular wastes. They consist of parallel troughs which alternate in depositing the material poured onto the splitter to the left or right. Manual mixing of a bucket of fly ash or foundry sand wastes, for example, was not adequate for the purposes of this study.

Liquid wastes can normally be mixed sufficiently to allow representative subsampling. In some cases, what appears to be a mixable liquid waste may have a sludge or immiscible layer at the bottom. It is always necessary to check for this with a stick, glass tube sampler, etc. In such cases it may be necessary to sample each layer separately, mixing them in the proper proportions for the leaching test.

Suspensions or solid-liquid mixtures can be very difficult to work with. If mixing does not provide uniformity, it may be necessary to sample solid and liquid portions separately, combining them for the leaching test, or even to keep them separate and run the test for each component, and mathematically synthesizing the results according to the original amounts of each material in the sample. This technique may also be necessary with some wastes containing solid components of different composition or leaching characteristics. It may be necessary to determine the amount of each distinct component present in the original waste, run the test for each component, and mathematically combine the results. The problem with running the leaching test separately for different components of a waste is that any interactions between components affecting the leaching patterns can unnaturally affect the results. This is not felt to be a problem with wastes arising from a single source, but could be important if different wastes are combined in the sample sent to the laboratory.

The test utilizes only the solid portion of the waste being studied. The liquid portion, namely that which will pass through a 0.45 micron filter, is analyzed directly for the components of interest. The rationale for the separation is that the liquid component of the waste can move away from the solid portion in the landfill, either due to gravity or capillary flow, or to absorption by surrounding materials. The liquid component of a waste represents an intrinsic potential impact on water quality as a result of waste disposal which is not dependent on external sources of water or leaching media. The solid portion remains behind for leaching, as simulated by the leaching test.

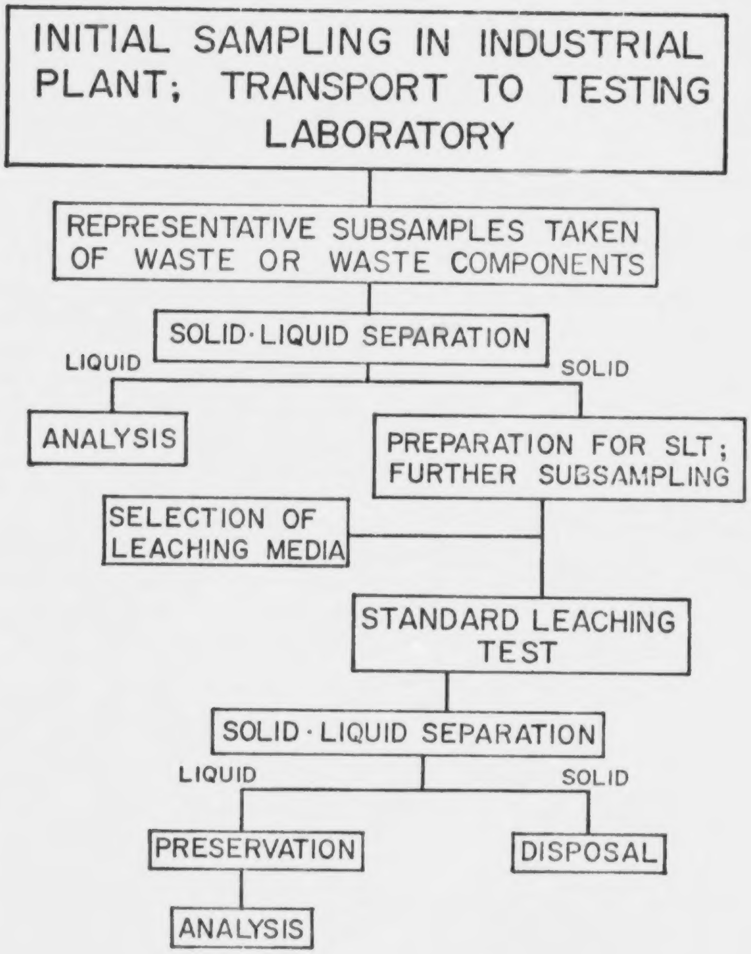

INITIAL SAMPLING IN INDUSTRIAL PLANT; TRANSPORT TO TESTING LABORATORY
REPRESENTATIVE SUBSAMPLES TAKEN OF WASTE OR WASTE COMPONENTS
SOLID·LIQUID SEPARATION
LIQUID
SOLID
ANALYSIS
PREPARATION FOR SLT; FURTHER SUBSAMPLING
SELECTION OF LEACHING MEDIA
STANDARD LEACHING TEST
SOLID·LIQUID SEPARATION
LIQUID
SOLID
PRESERVATION
DISPOSAL
ANALYSIS

Figure 49. Waste handling process.

In performing the solid-liquid separation, it is necessary to filter either the whole amount of waste in the sample, or to take a representative subsample, and perform the solid-liquid separation step using a 0.45 micron filter. Figure 9 (page 30, solid-liquid separation section) summarizes the recommended separation scheme which was found to be successful for all of the wastes tested. Note that centrifugation can be used if the filtration speed is too slow, and that the use of a pressure filter is also recommended if it is necessary to speed filtration. The final step is always the filtration of the liquid through a 0.45 micron filter. If it is not possible to perform a solid-liquid separation after using the steps mentioned in Figure 9, the residue on the filter is declared to be a solid for the purposes of subsequent testing. If the separated liquid portion of the waste is found to be hazardous, the test is terminated; otherwise the solids must be homogenized so they are representative and can be used in the leaching test. Bulky materials have to be reduced to smaller particles, e.g., 1 cm particle size.

The Leaching Test Procedure

The solids obtained from the sampling and solid-liquid separation processes are next leached in the following recommended procedure. First, the solid content of the solids has to be determined. This is because the leaching procedure calls for a set ratio of solids to leaching media based on dry weight solids. This is no problem with materials that are basically dry, such as fly ash, or with viscous sludges, etc., in which case the water content may be assumed negligible and the solids weight considered the dry weight. If the solids are highly absorptive, or if they contain volatile organics, problems develop. In the case of volatile organics, the dry weight, determined by drying to constant weight at 105°C, is continually changing depending on several factors including the volatility of the various components. Because the solid-liquid ratio is not as critical for testing purposes as it is for consistency and interpretation of the results, it is suggested that whatever weight is obtained after 24 hours drying at 105°C in a forced air convection oven be considered the dry weight for the purposes of the leaching test. Note that this "dry" weight, whatever it may be, can always be related back to the original waste sample as received, as a fraction or percentage of that sample, for purposes of interpretation.

Absorptive wastes present another problem. Wastes such as paper mill sludge or municipal refuse have a high absorptive capacity, so that, if such wastes are dried and mixed with the leaching media in the required proportions, they will soak up a major fraction of the leaching media, leaving little media available for analysis after solid-liquid contact. Further, subsequent elutions will have in effect a high solid-liquid ratio if the liquid is replaced by only as much leachate as was able to be separated in the case of procedure R, thereby slowing the release process. In procedure C, large amounts of waste and liquid must be handled to provide sufficient leachate for analysis, and replacing the waste in effect ties up or removes amounts of leaching media necessary to wet it. In effect, the solid-liquid ratios are not what was originally specified for test consistency with different wastes, and operational problems due to leaching media loss result.

It is recommended that absorptive wastes be dried to determine the dry weight, then an amount of wet waste necessary to meet the 1:10 solid-liquid ratio requirement (dry weight) be used in the various elutions. Note that the test results can still be related to the dry weight of the waste as received. Note also that the final composition of the liquid added to the leaching media in the form of liquid content of the waste (of known composition because it is analyzed earlier as immediately available) can be assumed equal to that removed at the end of each elution for analysis. Thus, this amount may be included in the total release obtained in the test. If the waste is very wet, so the liquid portion dilutes the leaching media to a significant degree, a more concentrated leaching media stock solution can be used, such that dilution by the liquid in the waste results in the normal concentrations. A correction curve to account for dilution of leaching media other than distilled water is included in the Appendix as Figure A-1.

Two elution procedures are used in the test, as shown in Figure 50; one in which the waste is replaced in each solution (Procedure C) and one in which the leaching media or eluent is replaced (Procedure R). Procedure C is designed to estimate the maximum concentrations of leachable species arising in the leachate as a result of leaching media contact with the solid portion of a waste, while Procedure R estimates the amounts of leachable species to be released. The latter may be expressed with most wastes on a weight of each constituent released per unit weight of waste basis, for example. The results of Procedure C are especially influenced by the waste, while Procedure R is more influenced by the leaching media. Both procedures use three 24-hour elutions with appropriate leaching media.

The choice of the leaching media used in the standard leaching test depends on the purposes of the test. Distilled water is to be used in all cases. It provides a data base for comparing different wastes, test procedures, and leaching media. Distilled water simulates rainwater and, so, it simulates the leaching capability of the waste when landfilled by itself or in situations where it controls leaching media composition. It also simulates contact with very old landfill leachate. As long as an industrial waste is landfilled in the open air, it will always be contacted for a certain time period with rainwater. Table 21 snows the relationship between the choice of the leaching media and the kind of landfill of interest for a specific waste.

The choice of the leading media is very important for it has a substantial effect on the results. The interpretation of the standard leaching test has to be done very carefully, taking into account the leaching media. The synthetic landfill leachate should be used if a certain waste is to be landfilled together with municipal waste. Directions for its preparation were presented in Table 14. It may be difficult to decide what leaching media should be used when the waste is to be landfilled with other kinds of industrial wastes which may control the leaching media composition, whether in an industrial or municipal landfill. Acid. base, or other leachates should be used, depending on the specific situation, or a leaching media obtained by prior contact of the other waste(s) and distilled water can be used. If the other wastes are biologically decomposable, a leaching media similar to the synthetic landfill leachate may be most appropraite.

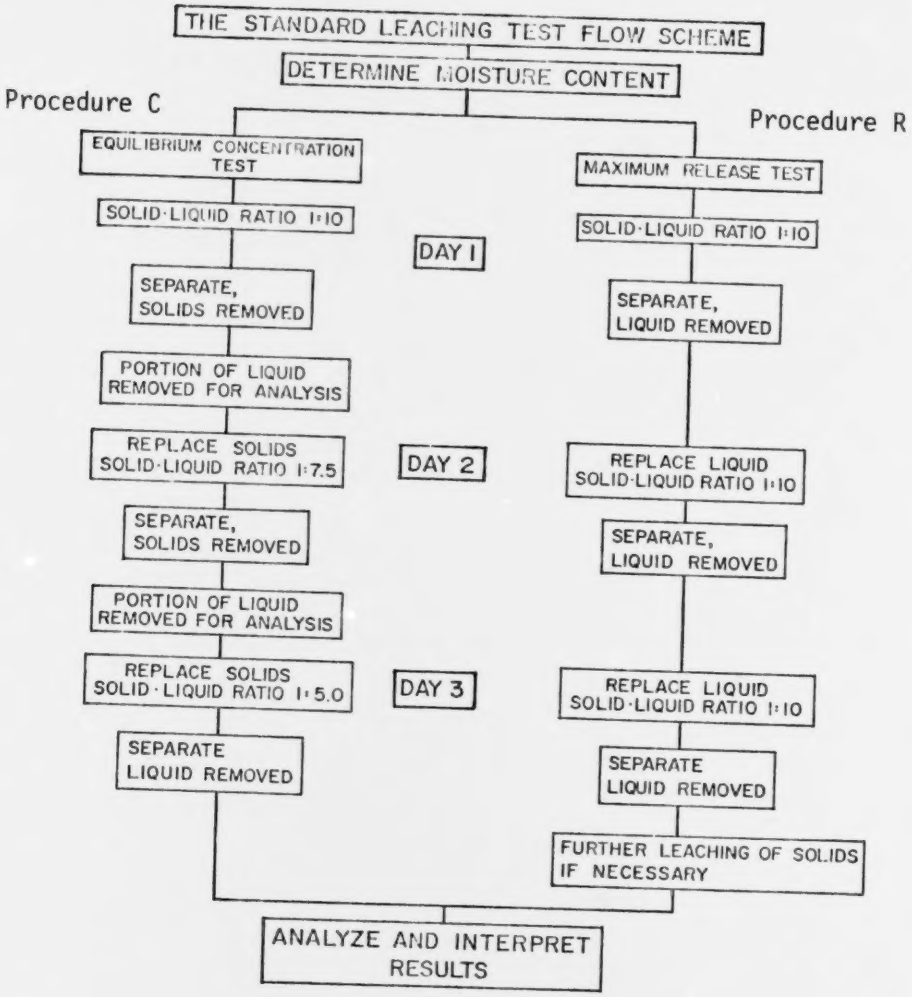

THE STANDARD LEACHING TEST FLOW SCHEME
DETERMINE MOISTURE CONTENT
Procedure C
Procedure R
EQUILIBRIUM CONCENTRATION TEST
MAXIMUM RELEASE TEST
SOLID-LIQUID RATIO 1:10
DAY 1
SOLID-LIQUID RATIO 1:10
SEPARATE, SOLIDS REMOVED
SEPARATE, LIQUID REMOVED
PORTION OF LIQUID REMOVED FOR ANALYSIS
REPLACE SOLIDS SOLID-LIQUID RATIO 1:7.5
DAY 2
REPLACE LIQUID SOLID-LIQUID RATIO 1:10
SEPARATE, SOLIDS REMOVED
SEPARATE, LIQUID REMOVED
PORTION OF LIQUID REMOVED FOR ANALYSIS
REPLACE SOLIDS SOLID-LIQUID RATIO 1:5.0
DAY 3
REPLACE LIQUID SOLID-LIQUID RATIO 1:10
SEPARATE LIQUID REMOVED
SEPARATE LIQUID REMOVED
FURTHER LEACHING OF SOLIDS IF NECESSARY
ANALYZE AND INTERPRET RESULTS

Figure 50. The recommended standard leaching test flow scheme.

TABLE 21. LEACHING MEDIA SELECTION ACCORDING TO LANDFILL CONDITIONS

Leaching Media	Landfill
Distilled water	to be used always, mono landfill
Synthetic leachate	industrial waste + municipal waste
0.1N H_2SO_4 } 0.1N NaOH }	municipal or industrial landfill; dependent on specific conditions
Others (organic solvents, liquids with high complexing capacity, etc.)	contacting the waste with other liquids in landfills containing other industrial wastes

In some cases the leaching test may be used to assess the effect on leaching of landfilling the waste being tested by itself or with other wastes. For example, one result might be that a particular waste should not be co-landfilled with acid wastes because the elution rate is extremely high.

For the first elution, both Procedures C (maximum concentration) and R (maximum release) use a 1:10 solid to liquid ratio (dry weight to volume). A separate sample of the waste is used to determine dry weight. In Procedure C, at the end of the first elution the sample is filtered, the solids discarded, and a portion of the leachate taken for analysis. An amount of fresh waste equal to that originally used (prewetted if necessary) is then added to the leachate. The amount of waste added is such that the solid to liquid ratio is 1:7.5 and 1:5 in the second and third elutions, respectively. The test is ended after the third elution. Procedure R uses fresh leachate on the same waste sample for each elution, thus maintaining a constant solid/liquid ratio of 1:10. More than three elutions may be used if there is reason for the additional elutions (such as variable leachate composition after each successive elution).

The test is run at ambient temperatures, unless special laboratory or landfill conditions dictate the use of another constant temperature, which should then be specially noted in the test records and reports. The test flasks should not be exposed to abnormal laboratory temperatures (sun, furnace, open windows, etc.).

A rotating mixer is used in conjunction with a square sample bottle. The mixer is tilted to give an almost-vertical rotational plane. As the bottles turn, the samples slide down the square sides and turn over somewhat in the process rather than simply staying at the bottom as would be the case in a round bottle. This agitation technique has been found to give good mixing with little or no abrasion. A rotation speed of 1 to 2 rpm is used. Square polyethylene containers are normally used, but glass flasks should be used when the plastic may affect the results. In general, plastic flasks are better if inorganic constituents in the leachate are of major concern, and glass flasks for organic constituents.

The amount of leachate and waste used is dependent on how much sample volume is needed for analysis, and in some cases on the minimum reasonable waste sample size. If only atomic absorption is to be used, for metal analysis, a sample volume of 100 ml is normally sufficient. That means for Procedure C at least 30 g of waste (dry weight basis) and 300 mls of leachate should be used. In Procedure R at least 10 g of waste (dry weight) and 100 mls of liquid would have to be used. It is recommended to use at least twice the above amounts, however, for some leachate is lost in the process of multiple elutions, or remains in the waste, filter, centrifuge tubes, glassware, etc. It is also recommended to choose the size of the square flasks according to the amount of leachate and waste required in order to fill up the flasks completely so that only a very small amount of air is in the flasks. This is especially important when the synthetic leachate is used so there is not enough oxygen trapped in the flasks to change the leachate composition significantly. If more than a few mls air space exists in a flask, it should be purged with nitrogen.

The test should be run with sufficient replicates to determine statistical reliability. Using sampling techniques described earlier, all of the wastes tested in this project gave reasonable reproducibility, and duplicates were sufficient. With very nonhomogeneous wastes, or wastes generally difficult to sample, a large sample size and more replicates will be required.

If gas production occurs in the flasks, the pressure should be released by using a pressure release valve (a spring-held ball valve worked satisfactorily), or by unscrewing the cap periodically until the gas is vented. After each elution, the solids and liquids are separated according to the same solid-liquid separation scheme used initially with the waste. Normally, 0.45 micron filtration occurs readily because of prior sample treatment and filtration. Filtration is necessary to obtain leachate for analysis and to replace either the solid or liquid fractions.

Leachate analyses are performed according to standard procedures. The selection of elements or compounds to be analyzed is waste dependent; however, in addition to specific species of interest because of the nature of each waste, or of interest for regulatory purposes, it is suggested to measure parameters that help in interpreting the results, such as pH, redox potential, specific conductance, COD, etc. If the leaching media used is synthetic leachate, some parameters cannot be measured accurately unless very high concentrations are leached, because these parameters are ingredients of the synthetic leachate. This includes Na, SO_4, Fe, and COD, plus the specific organics pyrogallol, acetate, and glycine.

During the test, observations should be written down such as change of the waste (color, particle size, appearance, odor, etc.), leachate (color, odor, etc.), gas production in the flasks, precipitation, etc. Such observations may be helpful in interpreting the results.

Presentation of the Results from the Standard Leaching Test

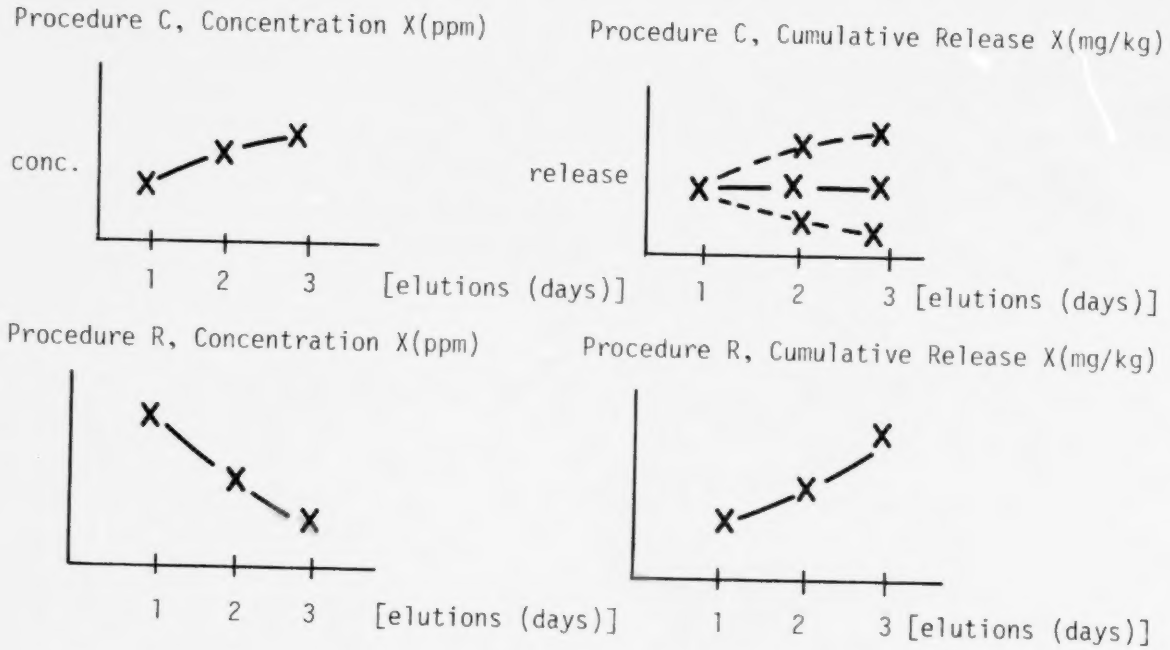
In this section some suggestions are made for summarizing and presenting the cumulative release and concentration data from the leaching test. Of course, the standard leaching test results from each waste have to be interpreted individually, but a standardized procedure for presenting the results may help to compare the leaching characteristics of different wastes and with different leaching media.

The results of the standard leaching test should be presented in the form of concentration and cumulative release curves for both Procedures C and R as shown in Figure 51. The cumulative release in mg/kg waste is based on the dry weight of the waste as used and defined in the standard leaching test procedures. Calculation of the cumulative release is performed by adding the product of the concentration of each chemical species and the volume of leachate obtained for each elutriation as shown in Table 22. For many wastes the curves will be similar to those presented in Figure 51. With Procedure C, the concentration increases with the number of elutions, for the waste is replaced after each elution and the liquid stays in the flasks. Thus, the leaching media is repeatedly contacted with fresh waste. The cumulative release curve for Procedure C may increase, stay constant, or decrease, depending on the slope of the concentration curve. The reason for these possibilities is that after each elution the waste is replaced by new non-leached waste. Therefore, the cumulative release after the second elution, for example, has to be related to the amount of waste used in both the first and second elutions.

With Procedure R, where the waste stays in the flasks for the whole test period and the liquid is replaced after each elution, the concentration will normally decrease with the number of elutions. As long as the concentrations of the second and subsequent elutions are not zero, the cumulative curve will increase.

There are situations in which the curves look quite different from those presented in Figure 51. This may happen, for example, with those parameters of solubility characteristics dependent on pH and with wastes which cause a pH change during the leaching test. In some cases the same concentration of some parameter is always reached in subsequent elutions using Procedure R. This means that the material is very soluble and there is a continuing supply of available material sufficient to approach solubility limits. The percentage of release of this specific parameter is constant with each elution. In this case, for a rough estimation of how many elutions may be necessary before the concentration drops, knowledge of the total amount of this parameter in the waste is useful.

The range in relative release rates is illustrated in Table 23 which presents the percentage release of four metals from papermill sludge (N), after three elutions. The composition of the papermill sludge was provided by the manufacturer and can only be assumed to be correct for the sample tested. It is interesting that the percentage release of the different elements is so different. The results suggest that Ca and Fe may leach from the papermill sludge for a long time, where K is very soluble and leaches at relatively higher rates for a shorter period. Mg may be in between. Explained differently, it can be expected that the K concentrations will decrease after a relatively short time while Ca and Fe will leach at approximately the same rate for a long time.



Procedure C, Concentration X(ppm)

Procedure C, Cumulative Release X(mg/kg)

Procedure R, Concentration X(ppm)

Procedure R, Cumulative Release X(mg/kg)

Figure 51. Suggested presentation of leaching test results for species X from a waste.

TABLE 22. CALCULATION OF CUMULATIVE RELEASE FOR PROCEDURES C AND R

Definitions:

C_i = concentration of leachate after ith elution, mg/ℓ.

ℓ_i = volume of leachate withdrawn after ith elution, ℓ.

w = weight of dry waste used in each elution, kg.

R_i = release from ith elution, mg/kg.

R = cumulative release, mg/kg.

Procedure C: (replacement of waste)

Elution 1: $\frac{C_1\ell_1}{w} = R_1$

Elution 2: $\frac{(C_2 - C_1)\ell_2}{w} = R_2 \qquad R = \frac{C_1\ell_1 + (C_2 - C_1)\ell_2}{2w}$

Elution 3: $\frac{(C_3 - C_2)\ell_3}{w} = R_3 \qquad R = \frac{C_1\ell_1 + (C_2 - C_1)\ell_2 + (C_3 - C_2)\ell_3}{3w}$

Procedure R: (replacement of leaching media)

Elution 1: $\frac{C_1\ell_1}{w} = R_1$

Elution 2: $\frac{C_2\ell_2}{w} = R_2 \qquad R = \frac{C_1\ell_1 + C_2\ell_2}{w} = R_1 + R_2$

Elution 3: $\frac{C_3\ell_3}{w} = R_3 \qquad R = \frac{C_1\ell_1 + C_2\ell_2 + C_3\ell_3}{w} = R_1 + R_2 + R_3$

where $\ell_1 = \ell_2 = \ell_3$.

TABLE 23. PERCENTAGE RELEASE RESULTS FOR PAPERMILL SLUDGE

Element	Original Concentration in Sludge [mg/kg]	Elution* [mg/kg]	% of Original Amount in Sludge Released
Ca	12,000	38	0.32
Fe	2,800	3.7	0.13
K	200	73	36.5
Mg	7,400	340	4.6

*After three days, based on dry weight, Procedure R.

It is often very time consuming and difficult, and in some cases impossible, to make a total digestion of a waste for the quantitative determination of its composition and to estimate total ultimate release. Accordingly, such a determination cannot be suggested as part of a standard leaching test, but it is recommended to obtain such information whenever it is critical or when it is not difficult to get the overall composition of the waste. It may be very helpful in interpreting the results.

A. Example of Presentation and Discussion of Results from a Standard Leaching Test

An example of the presentation of results from the leaching test procedure is shown in Figures 52 through 54 in which the copper oxide-sodium sulfate slurry waste was tested. For this particular test, a solid-liquid ratio of 1:7.5 (wet weight waste) happened to be used instead of 1:10 (based on dry weight). This does not affect the general pattern of the results. Distilled water and synthetic leachate were used as leaching media for 5 elutions. The maximum release (replacing the liquid) test Procedure R was done in duplicate. Figure 52 shows the influence of the test procedure and liquid on the pH values. Distilled water after contact with the waste had a pH in the 10 to 10.5 range. The pH values of the maximum concentration Procedure C are slightly higher, as would be expected. Especially when synthetic leachate was used, the conceptual differences between the two test procedures are obvious from the pH results. Procedure R pH was influenced by the leaching media pH and maintained acidic levels as the synthetic leachate buffered or controlled it at or near pH 4.5. With Procedure C, the buffering capacity of the synthetic leachate was overcome after the first elution and the pH was determined by the waste more so than by the leaching media. Note that the operational aspects of redox control were not perfected with the synthetic leachate at the time these results were obtained.

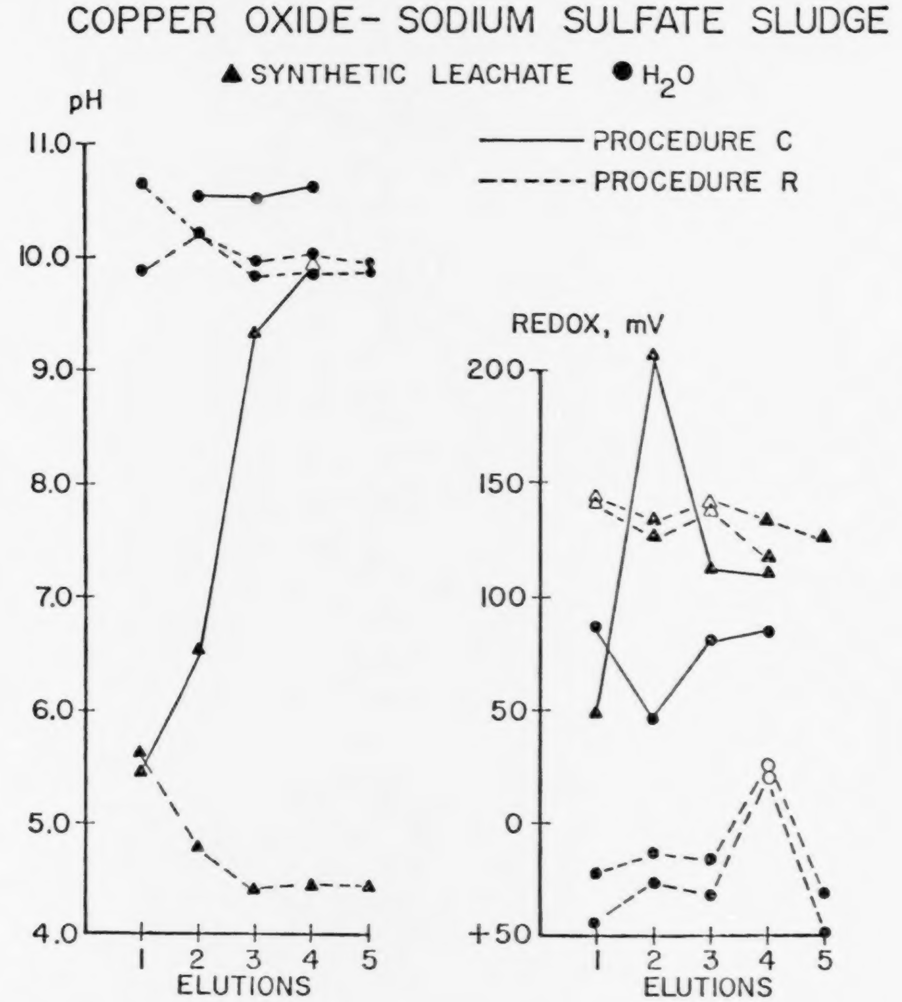


Figure 52. pH and redox from copper oxide-sodium sulfate sludge.

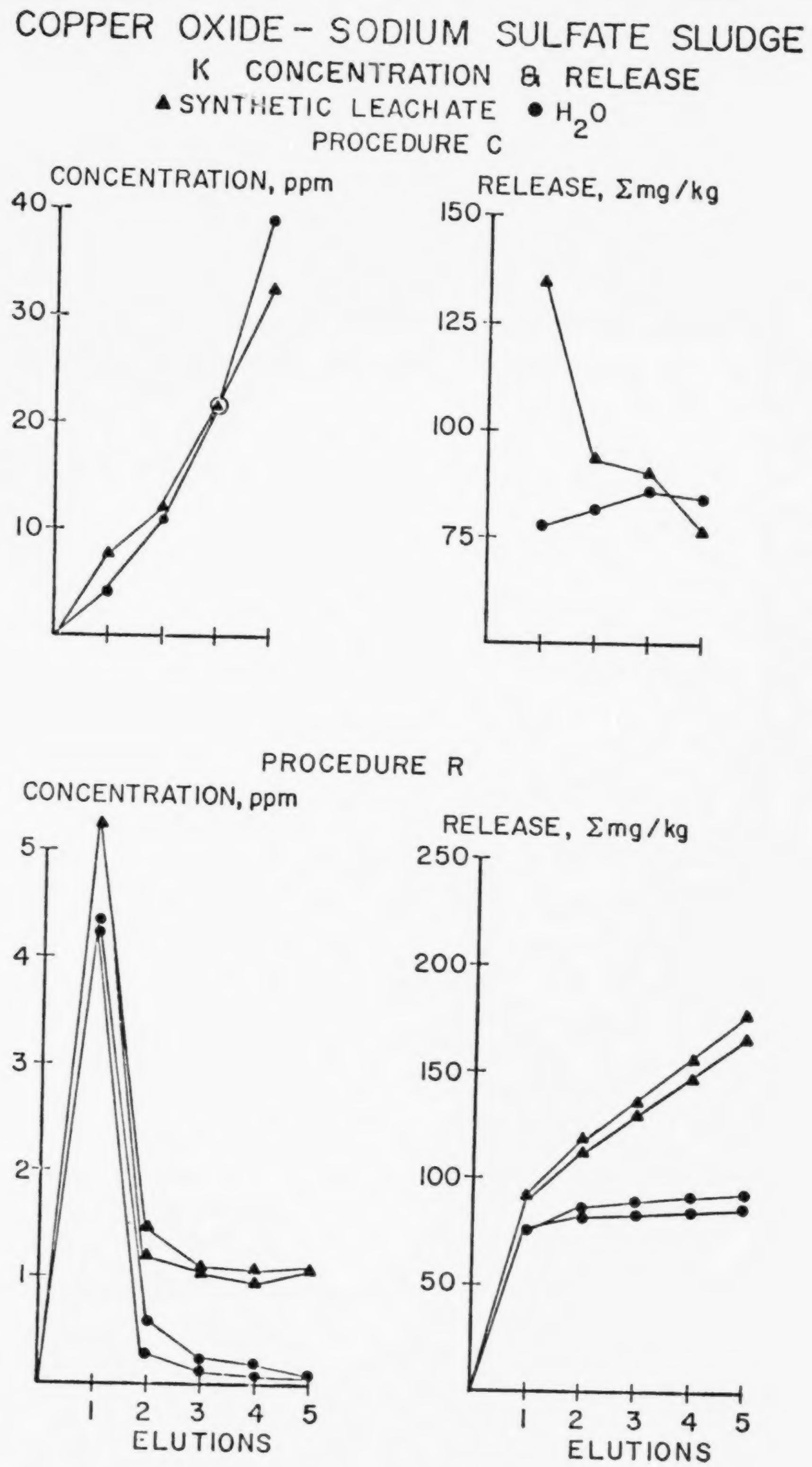


Figure 53. K concentration and release from copper oxide-sodium sulfate sludge.

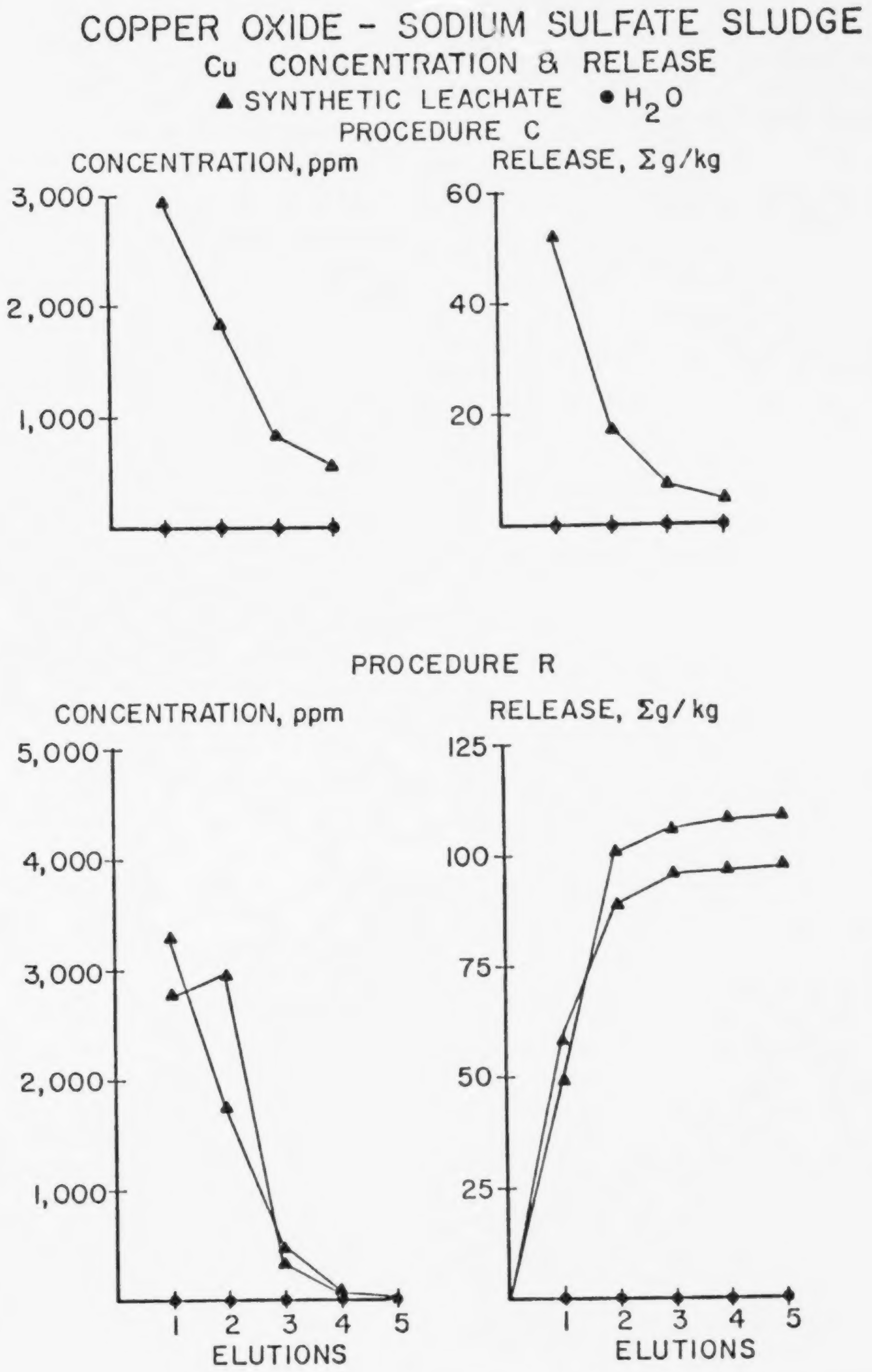


Figure 54. Cu concentration and release from copper oxide-sodium sulfate sludge.

The potassium concentration and release curves in Figure 53 are typical for chemical parameters for which release is not pH dependent. Procedure C shows increasing concentration levels as the number of elutions increases. The total release after 5 elutions using synthetic leaching media was approximately equal to that obtained using distilled water.

Compared with Procedure C, the results from Procedure R are very different. The concentration of potassium decreases with succeeding elutions in Procedure R, while the cumulative release increases. For this parameter, Procedure C shows highest concentrations with the tendency of further increases with more elutions. Procedure R indicates that the potassium dissolves quickly, and that the largest concentrations might be expected in the first phase of landfilling. The replacement of leachate in successive elutions in Procedure R resulted in more leaching-media dependency. The release with synthetic leachate was approximately double that using distilled water. After the large release from the first elution, the additional release (and concentrations) in subsequent elutions was basically constant, with synthetic leachate much more aggressive in leaching K than distilled water. This is probably due to the more rapid dissolution of the wastes by the synthetic leachate, bringing about an attendant release of K.

The results from both test procedures are different when the solubility of the measured parameters is pH dependent. An example of this situation is presented in Figure 54, where copper values are plotted. The use of distilled water as a leaching media resulted in no copper elution, whereas the concentrations resulting from use of synthetic leachate were extremely high. The use of distilled water only would have led in this case to unrealistic predictions of the leaching characteristics of this copper oxide-sodium sulfate slurry when co-landfilled with municipal refuse. The copper is very soluble in synthetic leachate, probably because of the acidic pH levels.

Of special interest is the decrease of copper concentration with the number of elutions. This occurred with both procedures, even though Procedure C is supposed to result in increasing concentrations to the maximum levels obtainable. In this case the basic nature of the waste nearly exhausted the buffering capacity of the synthetic leachate after the first elution and completely controlled the pH by the fourth elution. The solubility of copper, therefore, decreases with subsequent elutions as the pH rises. pH was not the only elution factor affecting Cu release, however, as shown by the results of Procedure C using synthetic leachate. Even though pH values of 9.3 and 9.9 were attained in the third and fourth elutions, at which levels the copper concentrations should have been negligible according to the results using distilled water as the leaching media, the complexing capacity of the synthetic leachate evidently resulted in the high Cu concentrations observed of 800 and 550 ppm.

If it is assumed that Cu, Mg, and Na are the only major cations of the copper oxide-sodium sulfate slurry waste, and exist in the compounds CuO, $MgSO_4$, and Na_2SO_4, basically 100% of these materials were leached in the five elutions in Procedure R using synthetic leachate as a leaching media. Na values used for this calculation were those measured in distilled water because of the high background level of Na in the synthetic leachate. The calculated percentages are 99.43 and 107.5% for the two duplicate runs.

The results of the leaching test in which the copper oxide-sodium sulfate slurry was used are not typical of all wastes, because this waste is very soluble under the described conditions. This example was chosen because it shows very clearly the effect of the synthetic leachate and the two test Procedures C and R on the waste's leaching characteristics. The interpretation of the test results illustrates the benefit of using both procedures and leaching media in the leaching test.

Absolute meaningful interpretation of the results from the leaching test to describe the potential environmental hazard of the waste is very difficult to do. One observation that is clear is that the described results of the copper oxide-sodium sulfate slurry indicate that this waste should not be landfilled together with municipal waste unless special precautions are taken, because the release of copper is very high in this situation.

Interpretation of Leaching Test Results

It is clear that no finite laboratory leaching test can predict with certainty the long term leaching pattern of a waste in a landfill. A compromise must be made in selecting either a batch (flask) test or a column test, and, once that basic decision is made, a multitude of conceptual and operational decisions must be made in specifying exact test procedures. The alternative of leaving such decisions for the laboratory technician to make for each waste, as seems appropriate, eliminates standardization of procedures and direct comparison of the results from one laboratory to another, from one waste to another, etc. Additional compromises have been made and operational variables and techniques have been evaluated and specified as recommended as part of this study, but the test results are, at best, difficult to apply to full-scale landfill situations.

Numerous test conditions were studied as part of this research. Such variables as choice of leaching media, solid-liquid ratio, number of elutions, time per elution, agitation technique, temperature, surface area of contact, biological effects, sampling and sample preparation, solid-liquid separation, and others were considered. Of the leaching test conditions, leachate composition and temperature are most readily related to landfill conditions. Other test conditions are more difficult to base on landfill conditions and were chosen based on criteria other than landfill modeling. Since these conditions can affect a parameter's concentration in the test leachate, caution must be exercised in data interpretation. One cannot always transpose test data to landfill practice directly.

It may be possible to correlate test conditions with landfill concentrations by running extensive verification tests, correlating a waste's behavior in the test with the behavior of the same waste in a carefully monitored landfill. Correlation coefficients could then be developed for parameters and conditions similar to those in the verification study, and the test result used to estimate landfill concentrations.

A short leaching test cannot duplicate completely the long-term leaching characteristics of a waste. The short test might miss completely a parameter having exponential release or might overestimate the release of a parameter showing a concentration maxima. These patterns could be seen in a long-term study, but are difficult to determine in a short test.

When interpreting test results, it is important to consider the physical condition of the landfilled waste. Of special concern in this regard would be a waste which is landfilled in large stable chunks or with a stable impervious coating which could behave far differently in a landfill than in a test in which it were ground. Although not specifically tested as part of this study, it seems reasonable to cut, crush or specially make such wastes to yield particles approximating the size equivalent to a one cm cube, for example. This particle size is small enough to work in the procedure, yet large enough to not increase drastically the surface area per unit waste exposed to leaching. Interpretation of results may involve use of a factor compensating for the increased surface area obtained because of size reduction techniques. A simple ratio between the most likely surface area of the waste in the landfill per unit weight to the surface area per unit weight for the particle size as tested could be used to adjust at least the initial results for cumulative release. If subsequent elutions indicate negligible particle breakdown or secondary release with time, such a technique should be adequate.

An evaluation of the hazardous nature of a waste must incorporate an evaluation of the waste's landfill environment. The hazardous nature of a waste is a siutation-specific characteristic. For example, a waste may be hazardous to an organism under one set of environmental conditions, yet completely innocuous under a different set of conditions. Furthermore, its hazard may be organism specific; i.e., it may be hazardous to one organism and not to another under the same set of conditions. Thus, a determination of the hazardous nature of a waste must include an evaluation of its effect on specific organisms (or plants or animals, etc.) under specific conditions.

Some appreciation for the release and concentration levels obtained may be obtained by comparing results from the leaching test of the waste in question with various natural materials. For example, the leaching test was run on park soil, municipal waste (City of Madison) and dried digested sewage sludge (from the Madison Metropolitan Sewage District Nine Springs Sewage Treatment Plant) in order to generate background data on leaching of more or less natural materials. These results are shown in Tables 24 and 25. It is clear that some materials which are

TABLE 24. MAXIMUM CONCENTRATIONS AFTER THREE ELUTIONS OF SOIL, MUNICIPAL REFUSE AND SEWAGE SLUDGE IN mg/ℓ, PROCEDURE C

Substance	Leaching Media	Cu	Zn	Fe	Mg	Pb	Na	K
Soil	Dist. H_2O	0.03	0.02(0.04)	0.15(2.6)	17	b.d.l.	162	16.2
	Syn. leach.	----	----	----	----	-----	----	----
	0.1 N H_2SO_4	0.11(0.44)	2.43(3.87)	3.4(99.6)	372(429)	1.1(1.7)	838	46.2
Municipal Refuse	Dist. H_2O	3.24	21.1	22.5	109	1.4	1680	306
	Syn. leach.	2.75(4.0)	47	----	131	6.9	----	306
	0.1 N H_2SO_4	3.83(5.3)	25.9	72.9	247	1.4(2.66)	1430	226
Dried Digested Sewage Sludge	Dist. H_2O	0.09	0.58	4.95	146	b.d.l.	203	69.3
	Syn. leach.	0.9(1.02)	26.5(29)	----	190	b.d.l.	----	64.8
	0.1 N H_2SO_4	0.39(3.1)	36	1.2(105)	210(296)	(2.65)	219	71.1

b.d.l = below detection limits.

Note: Number in parentheses signifies maximum concentration measured if not attained after three elutions.

TABLE 25. CUMULATIVE RELEASE AFTER THREE ELUTIONS OF SOIL, MUNICIPAL REFUSE, AND SEWAGE SLUDGE IN mg/kg DRY WASTE, PROCEDURE R

Substance	Leaching Media	Cu	Zn	Fe	Mg	Pb	Na	K
Soil	Dist. H_2O	0.3	0.7	27.8	109.8	b.d.l.	3430	144
	Syn. leach.	2.3	54.2	----	2720	22.5	----	515
	0.1 N H_2SO_4	6.4	67.5	1140	8900	33.8	3300	440
Municipal Waste	Dist. H_2O	12.8	73.6	94.2	479	b.d.l.	5060	1510
	Syn. leach.	122	338	----	560	105	----	7580
	0.1 N H_2SO_4	630	470	1990	4190	67.1	5250	1710
Dried Digested Sewage Sludge	Dist. H_2O	0.7	2.1	7.3	347	b.d.l.	1787	194
	Syn. leach.	20.5	606	----	1590	b.d.l.	----	604
	0.1 N H_2SO_4	39.7	811	1110	6180	26.5	2310	456

b.d.l. = below detection limits.

regularly deposited on soil, or exist in soil, can be released via a leaching process in amounts comparable to some industrial wastes. It may also be that the natural soils at a particular location have a beneficial effect in attenuating or otherwise changing contaminant release from a waste. Perhaps a test procedure would be useful in which samples of the soils of interest at a specific site are added in varying amounts to the leaching test flask to better predict any relationships that might occur between soil and the waste in a landfill.

One obvious way to interpret the leachate composition results is to compare the concentrations of the various chemical species to some standard, for example drinking water standards. This is dangerous, however, and is difficult to defend for the leaching test developed in this study. Obviously, test requirements could have been adjusted to yield virtually any concentration of leachable species in the leachate for analysis. This could have been done, for example, by use of different elution times (e.g., 5 minutes instead of 24 hours), or solid-liquid ratios. The test was designed to be rapid, aggressive, and to yield as much information about the leaching characteristics of a waste as possible in a relatively short time. It was not designed to provide realistic concentrations of the various species for a specific situation.

One method for interpretation of test results involves adjustment of measured chemical species concentrations by a factor. This factor may be based in part on the amount of leaching media the waste might contact during the estimated active life of the landfill. Thus, the cumulative infiltration expected over a ten-year period at an appropriate landfill might be used. For example, if a waste is found to release a total of 0.1 gram of a chemical species of concern S per kilogram of waste in the leaching test, and the waste is to be landfilled in an area that receives an average of 100 cm precipitation per year of which 25% is net infiltration and results in leaching, the following calculation can be made. The average concentration of S over an assumed active leaching life of 10 years from one ton of waste is $\overline{C}$.

$$1000 \text{ kg waste} \times \frac{0.1 \text{ g species S}}{\text{kg waste}} = 0.1 \text{ kg species S evolved}$$

Assume the waste is landfilled so that 1 meter^2 landfill area holds 1 ton waste.

$$\overline{C} = \frac{0.1 \text{ kg S} \times 10^6 \text{ mg/kg}}{10 \text{ years} \times 1.0 \frac{\text{m pptn.}}{\text{yr}} \times 0.25(\text{fract. infilt.}) \times 1 \text{ m}^2(\text{surface area}) \times 1000 \text{ l/m}^3}$$

$$= 40 \frac{\text{mg S}}{\text{l}}$$

The selection of numbers may be unrealistic (for example, the use of one meter thick landfill), but the example illustrates an attempt to relate average release data to landfill conditions. The above calculation could be further modified by the amount of waste expected to be landfilled per unit landfill area.

Another way of developing interpretive criteria when toxic effects are of primary concern would be to dilute both the leaching media and the elutriates obtained by waste contact to the point where no toxic effects are noted in a toxicity test. The difference in dilutions required to provide no toxic effects could be a measure of the incremental effect of the leachable constituents of a waste on the leaching media, and would, thus, be an indication of the hazardous character of the waste. The difference would have to be interpreting knowing that the test is designed to be aggressive, but at least the various base-level degrees of toxicity inherent in the selection of the raw leaching media are taken into account.

In summary, the standard test provides a rapid indication of which chemical species are immediately leached from a waste, and an indication of the maximum concentrations of each specie likely to be found in the leachate. In addition, an estimate of the amount of each specie likely to be released per unit weight of waste is obtained. Finally, the test can provide valuable information with regard to the relative effect of co-disposal of the waste in question with other wastes or mixed municipal refuse. The fact that the test is not perfect in predicting the long-term leaching pattern of a waste, or the precise concentration of a particular parameter in a particular landfill, for example, means that the test results need to be interpreted with care.

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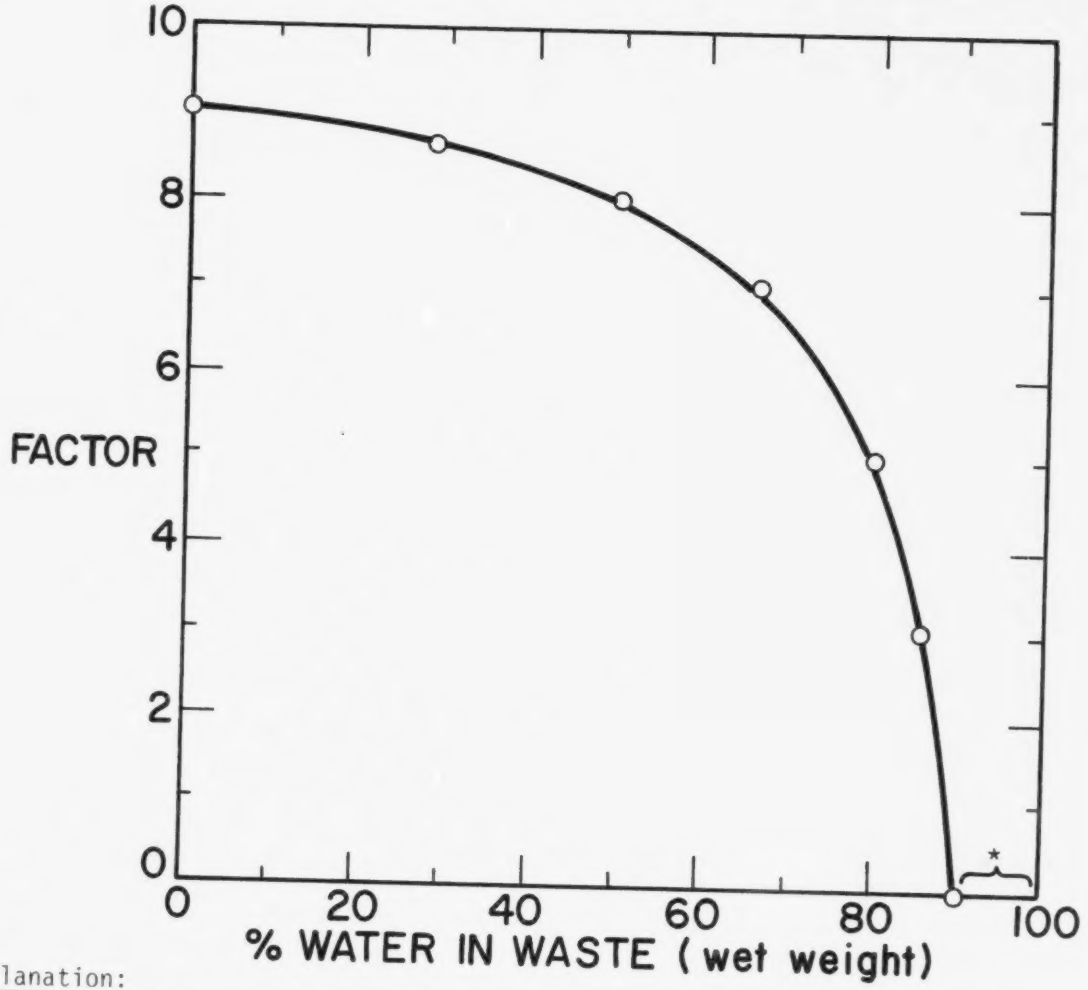
APPENDIX



Explanation:

1) Concept is to use ten times normal strength synthetic leachate concentrate, reducing the amount of distilled water to dilute the concentrate by the amount of moisture in the waste.
2) Grams dry waste = ml. of 10x conc. synthetic leachate.
3) Volume (mls) of distilled water to dilute cons. (10x) synthetic leachate = volume conc. (10x) synthetic leachate times the factor plotted above.
4) Example: Desired mix = 60 gr. dry waste and 600 ml. synthetic leachate media (1:10 ratio). Amount 10x concentrated synthetic leachate = 60 mls; amount distilled water for wet waste at 50% moisture content = 60 × 8 = 480 mls (8 is factor plotted above).
5) For Procedure R, replacing the waste, add an amount in ml. of 10x conc. synthetic leachate equal to 1/9 the weight of moisture added in the wet waste each time waste is added. Continuing example (4 above), if waste is replaced for elution 2, add 60/9 or 6.7 ml. 10x conc. synthetic leachate to counter the dilution by the moisture in the wet waste.

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*For % water in waste > 90%, use required amount wet waste plus mls. 10x conc. synthetic leachate = grams dry waste only.

Figure A-1. Correction of the moisture content when using the synthetic leachate.

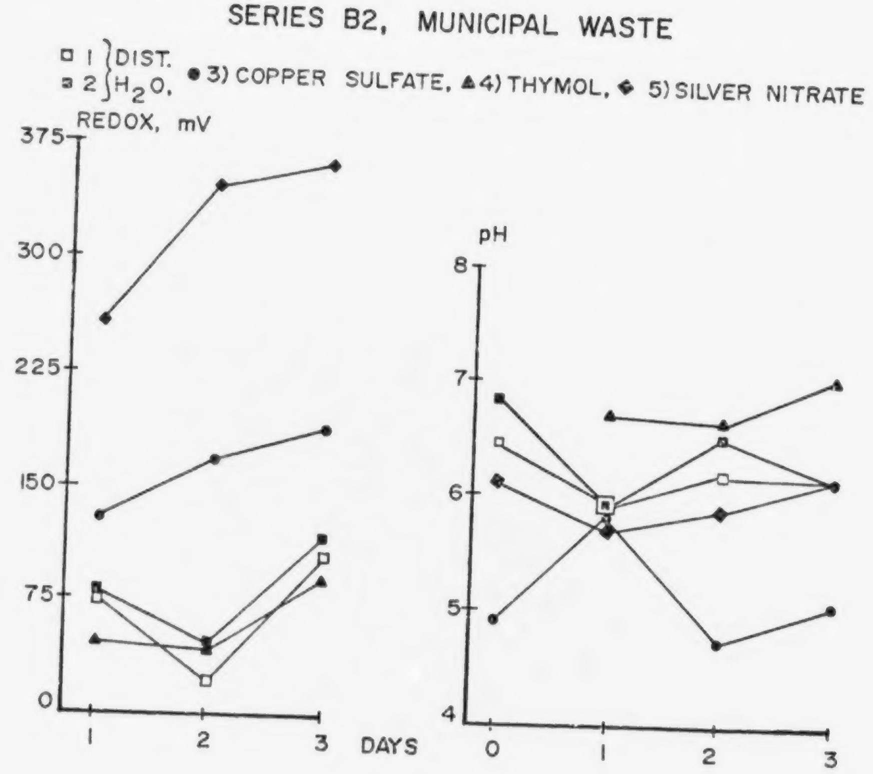

Figure A-2. Test B2 on the effects of various biologically inhibiting agents on leaching of municipal wastes. (See text for procedure.) Redox and pH.

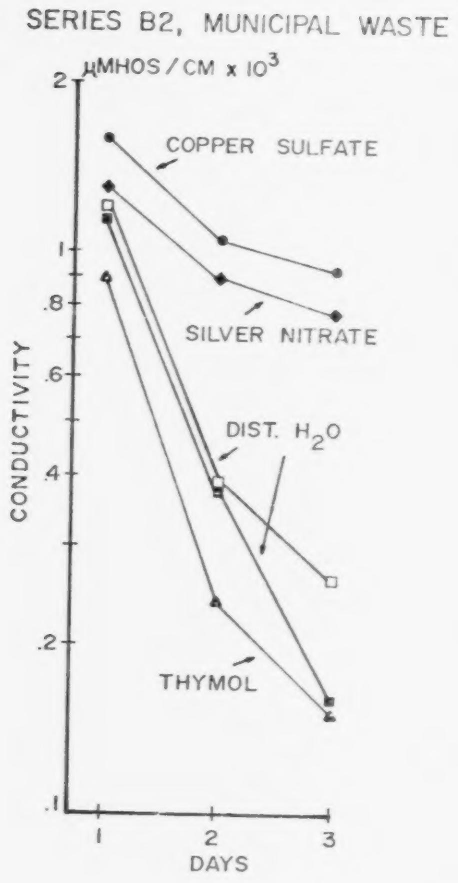

Figure A-3. Test B2 on the effects of various biologically inhibiting agents on leaching of municipal wastes. Specific conductance.

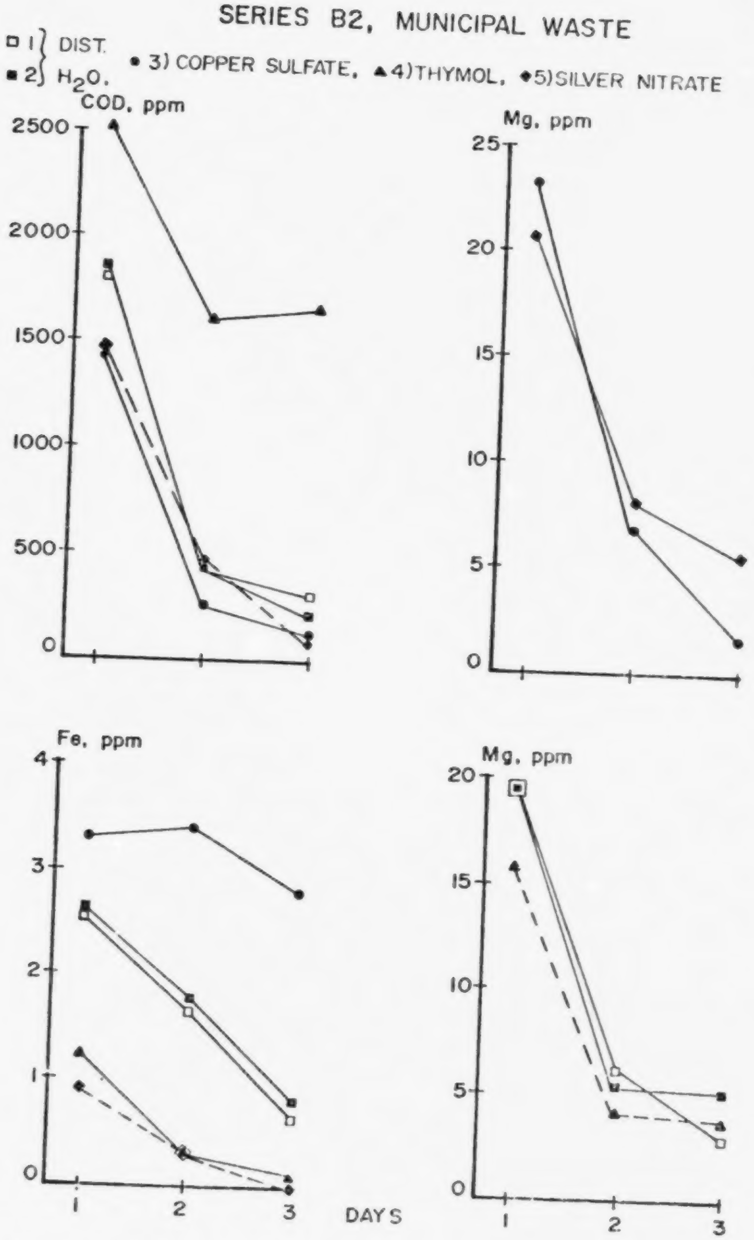

Figure A-4. Test B2 on the effects of various biologically inhibiting agents on leaching of municipal wastes. COD, Mg, and Fe.

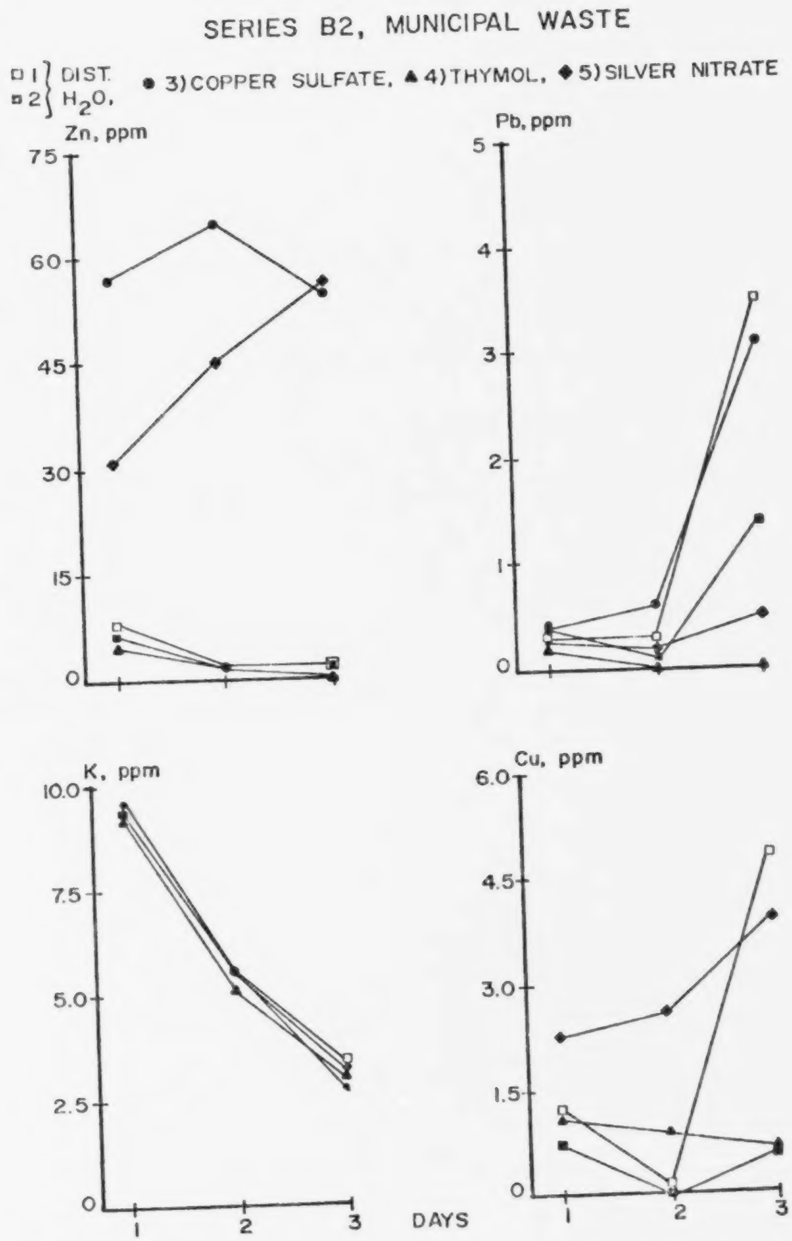

Figure A-5. Test B2 on the effects of various biologically inhibiting agents on leaching of municipal wastes. Zn, Pb, K, Cu.

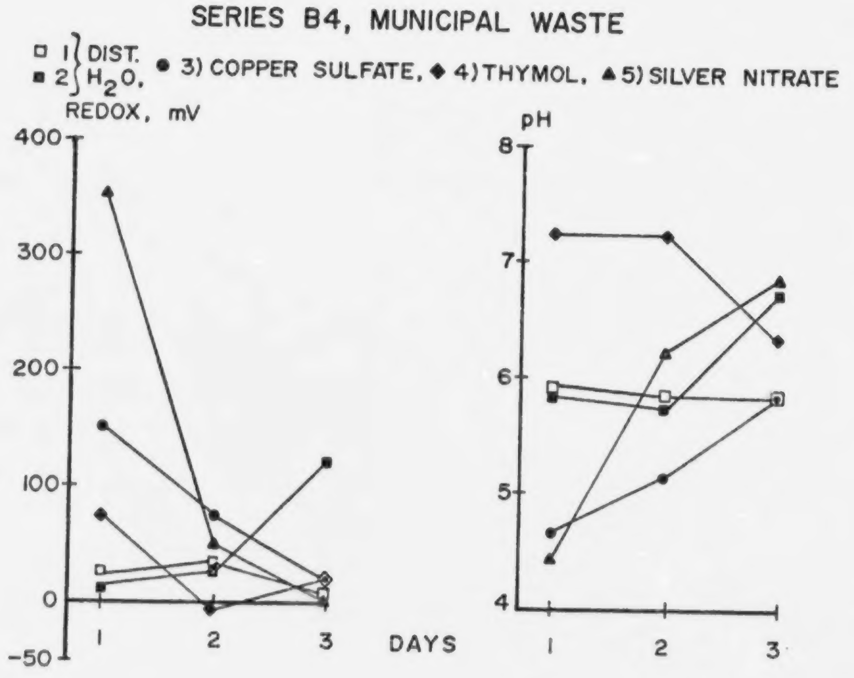

Figure A-6. Test B4 on the effects of various biologically inhibiting agents on leaching of municipal wastes. (See text for procedure.) Redox and pH.

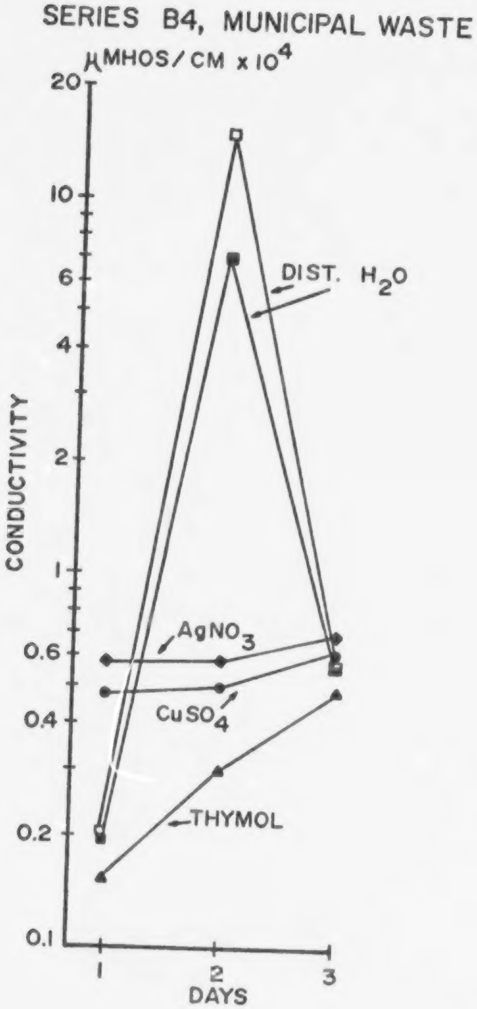

Figure A-7. Test B4 on the effects of various biologically inhibiting agents on leaching of municipal wastes. Specific conductance.

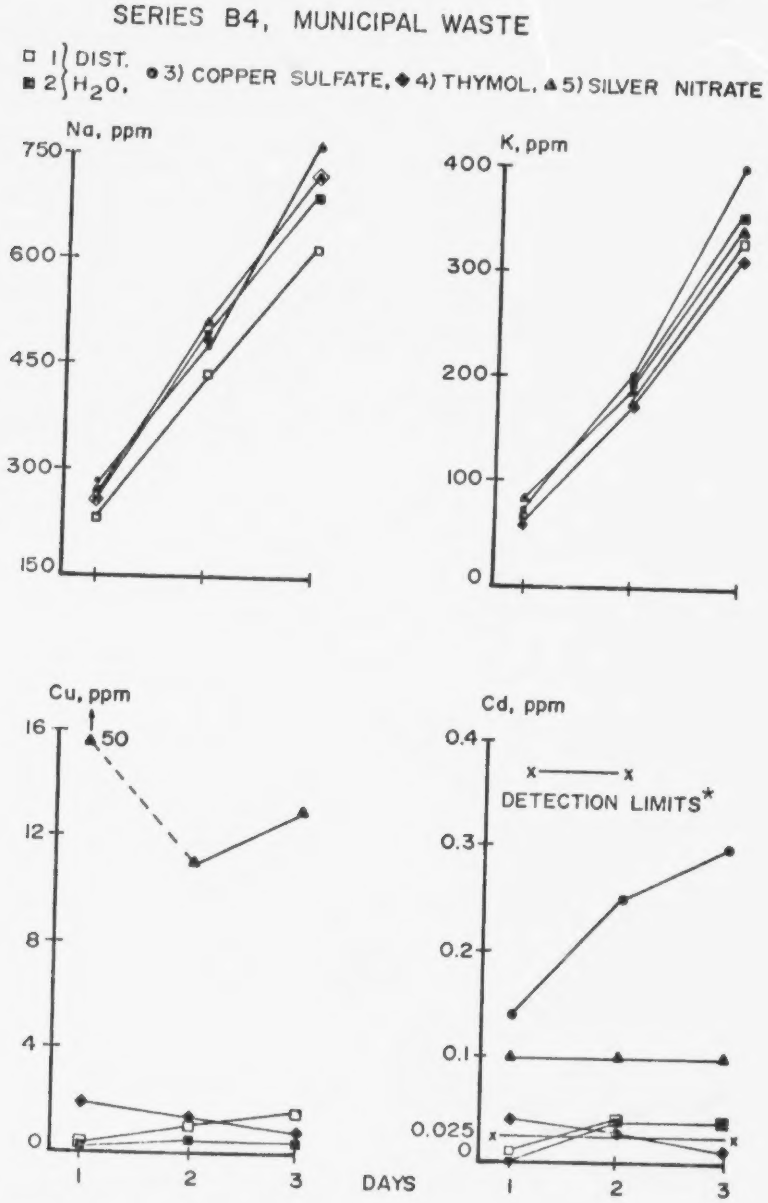

Figure A-8. Test B4 on the effects of various biologically inhibiting agents on leaching of municipal wastes. Na, K, Cu, Cd.

*All points below detection limit are approximate.

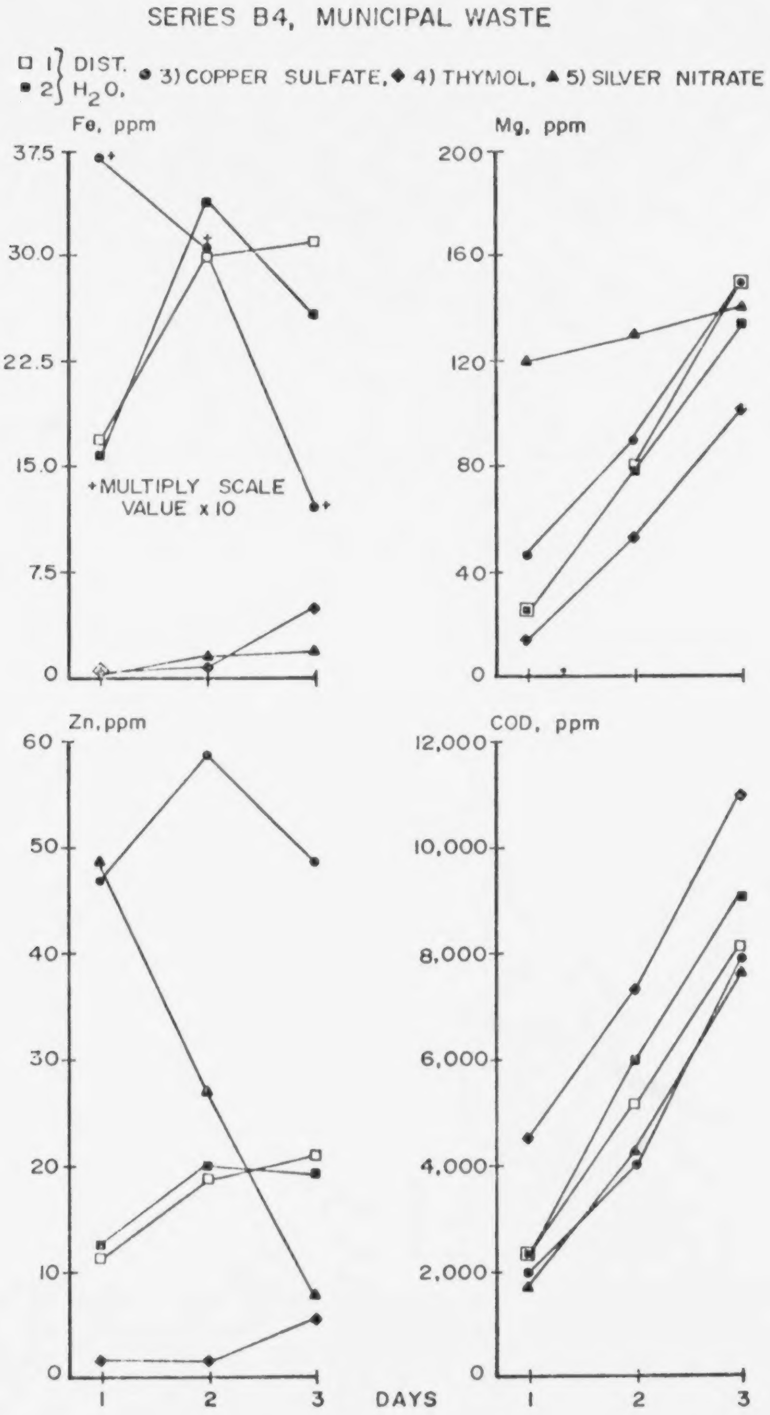

Figure A-9. Test B4 on the effects of various biologically inhibiting agents on leaching of municipal wastes. Fe, Mg, Zn, COD.

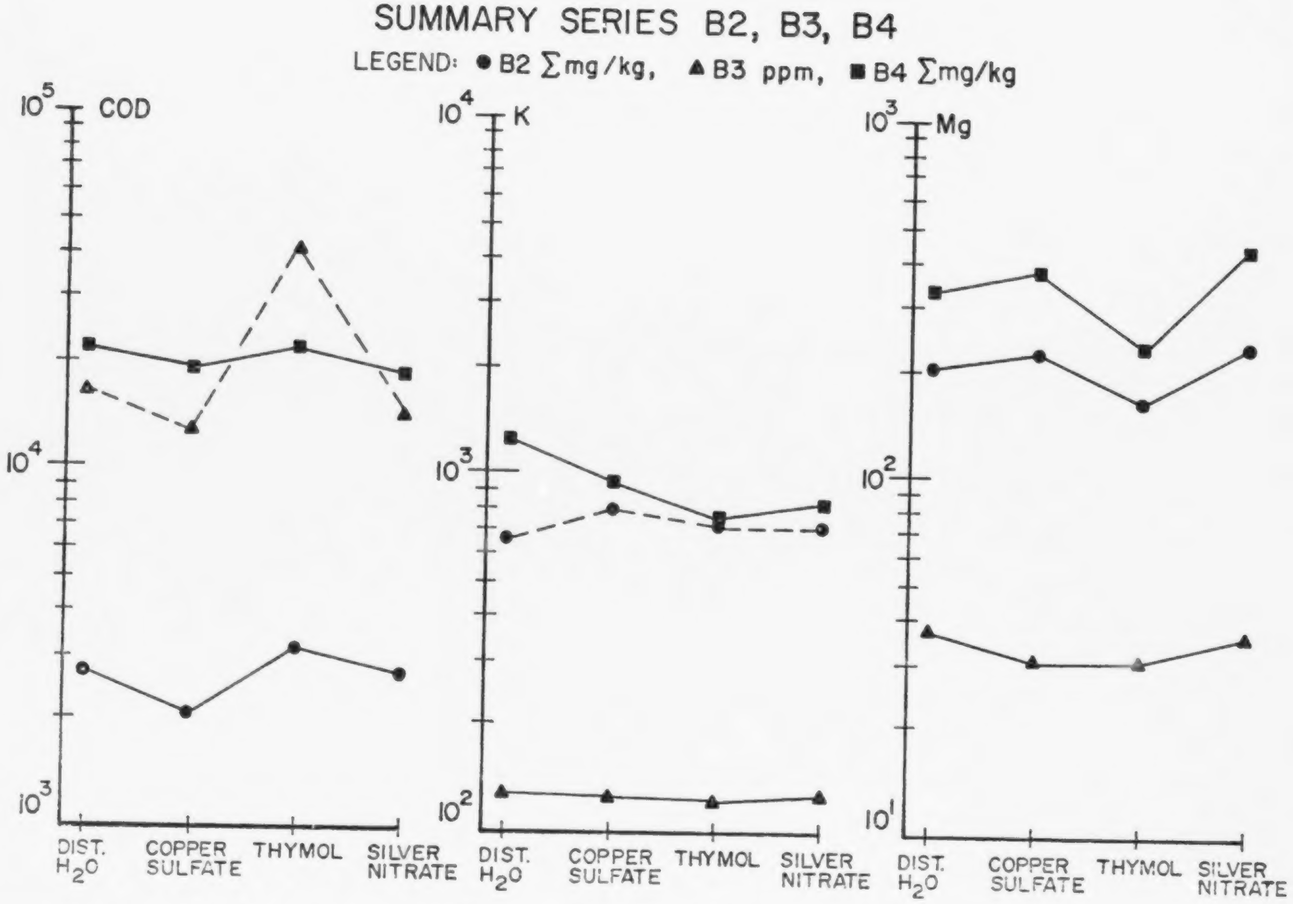

Figure A-10. Comparison of tests B2, B3, and B4. (See text for procedure.) COD, K, Mg.

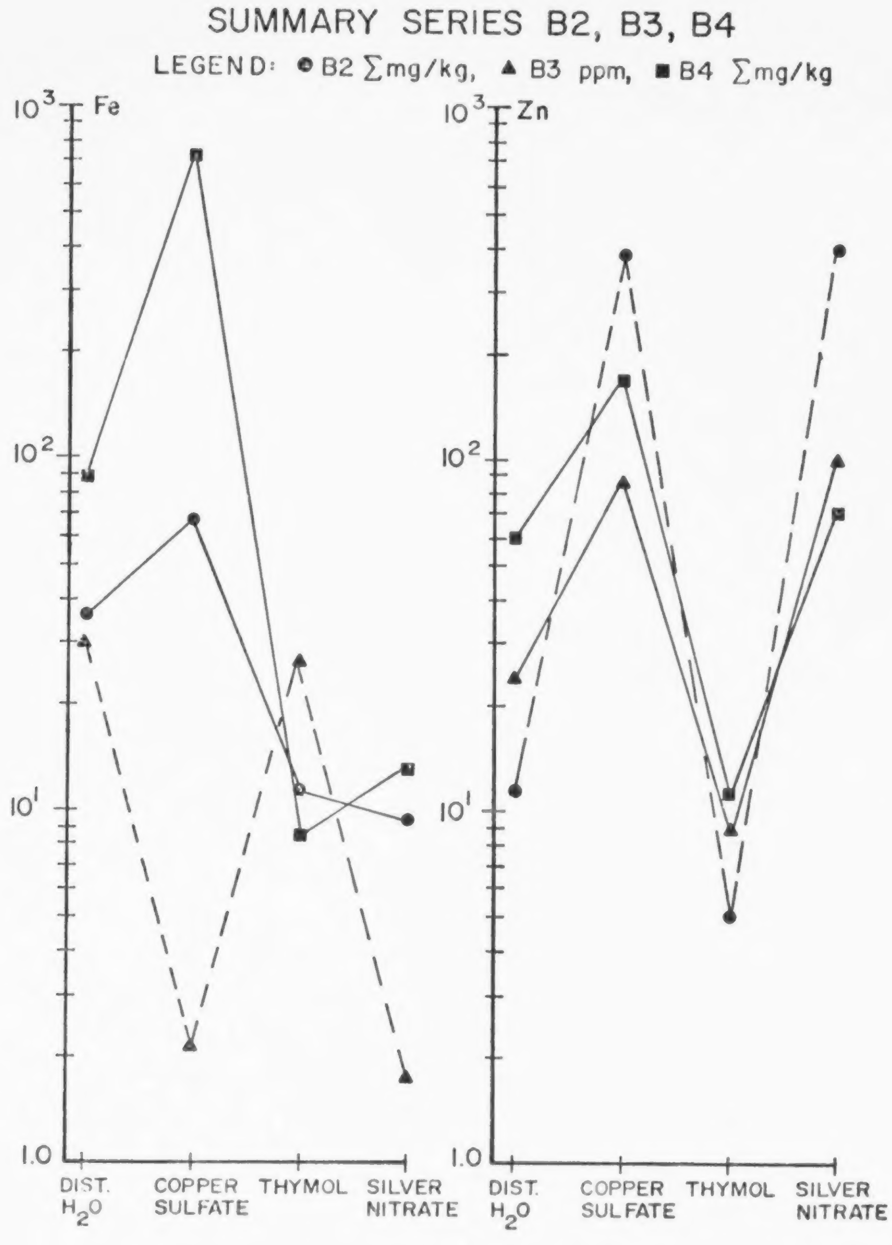

Figure A-11. Comparison of tests B2, B3, and B4. (See text for procedure.) Fe, Zn.

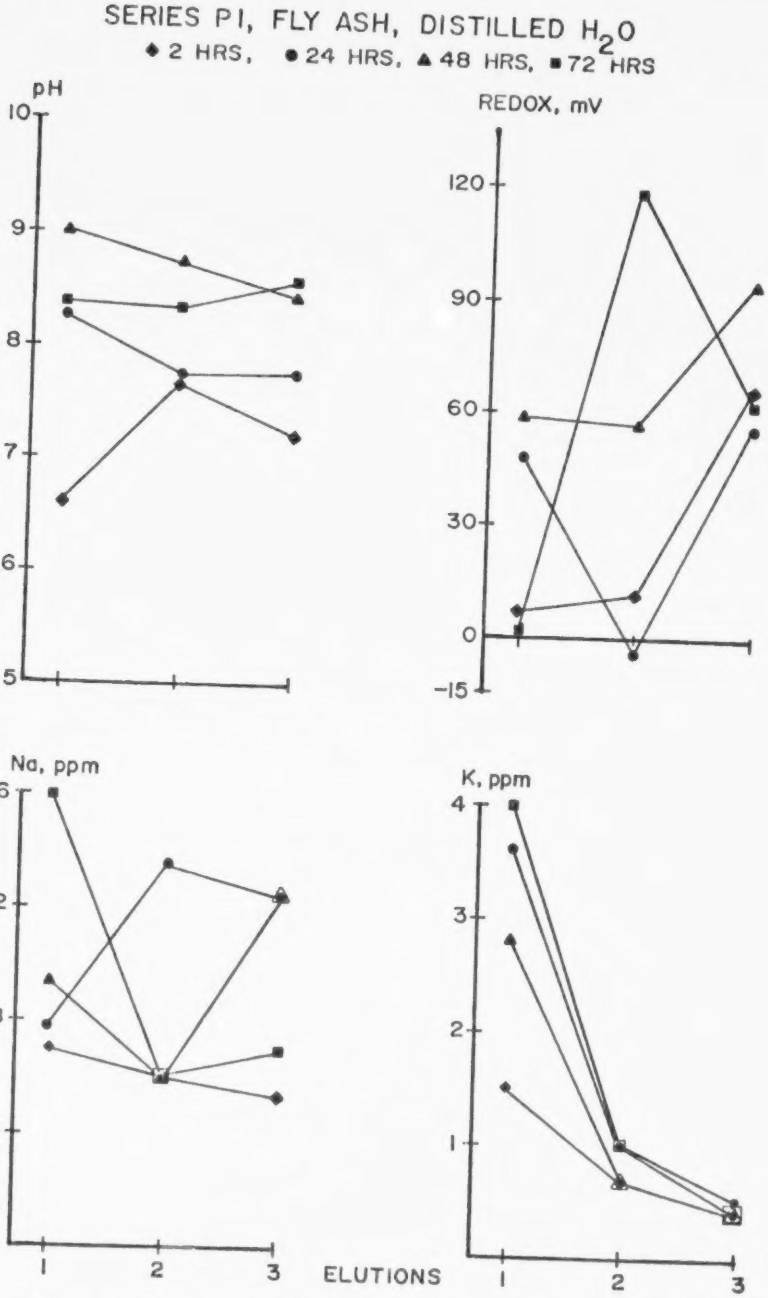

Figure A-12. Test P1 on the effect of time per elution using procedure R on fly ash with distilled water. (See text for procedure.) pH, Redox, Na, K.

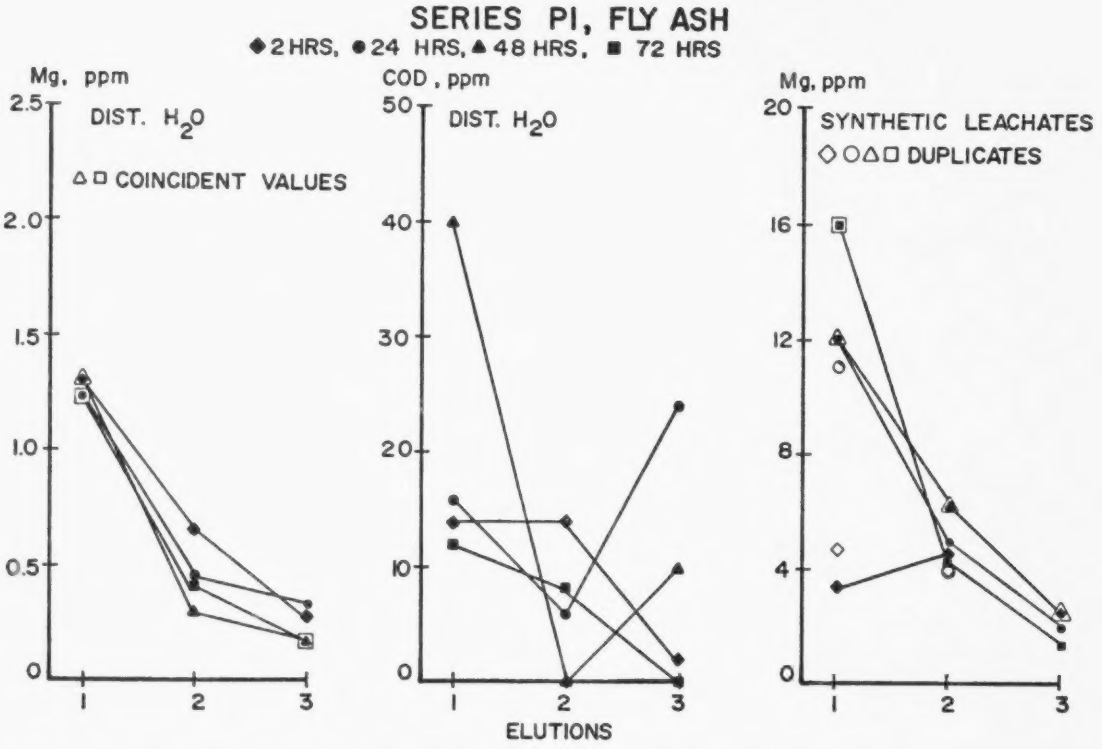

Figure A-13. Test P1 with fly ash and distilled water (Mg and COD) and synthetic leachate (Mg).

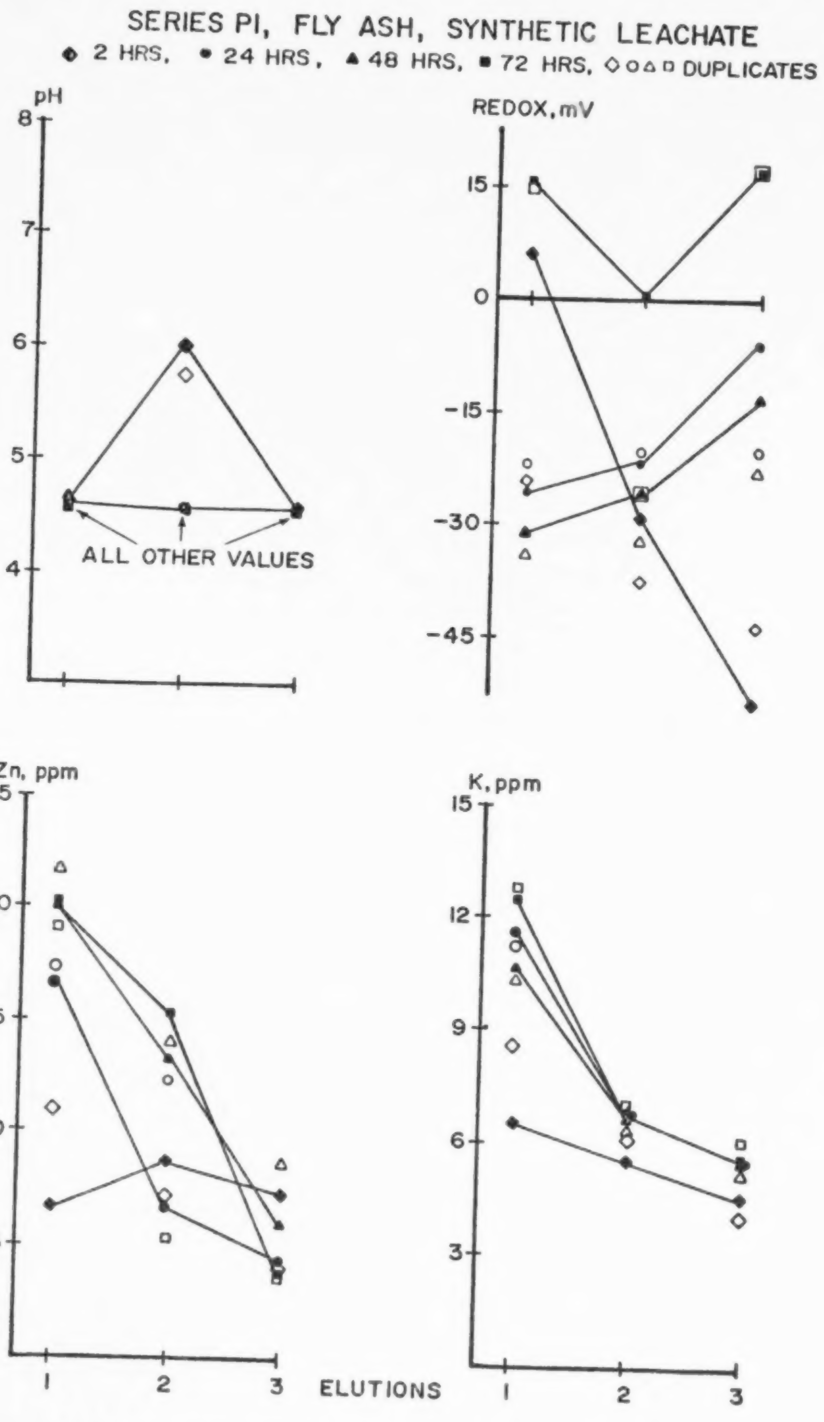

Figure A-14. Test P1 with fly ash and synthetic leachate. pH, Redox, Zn, and K.

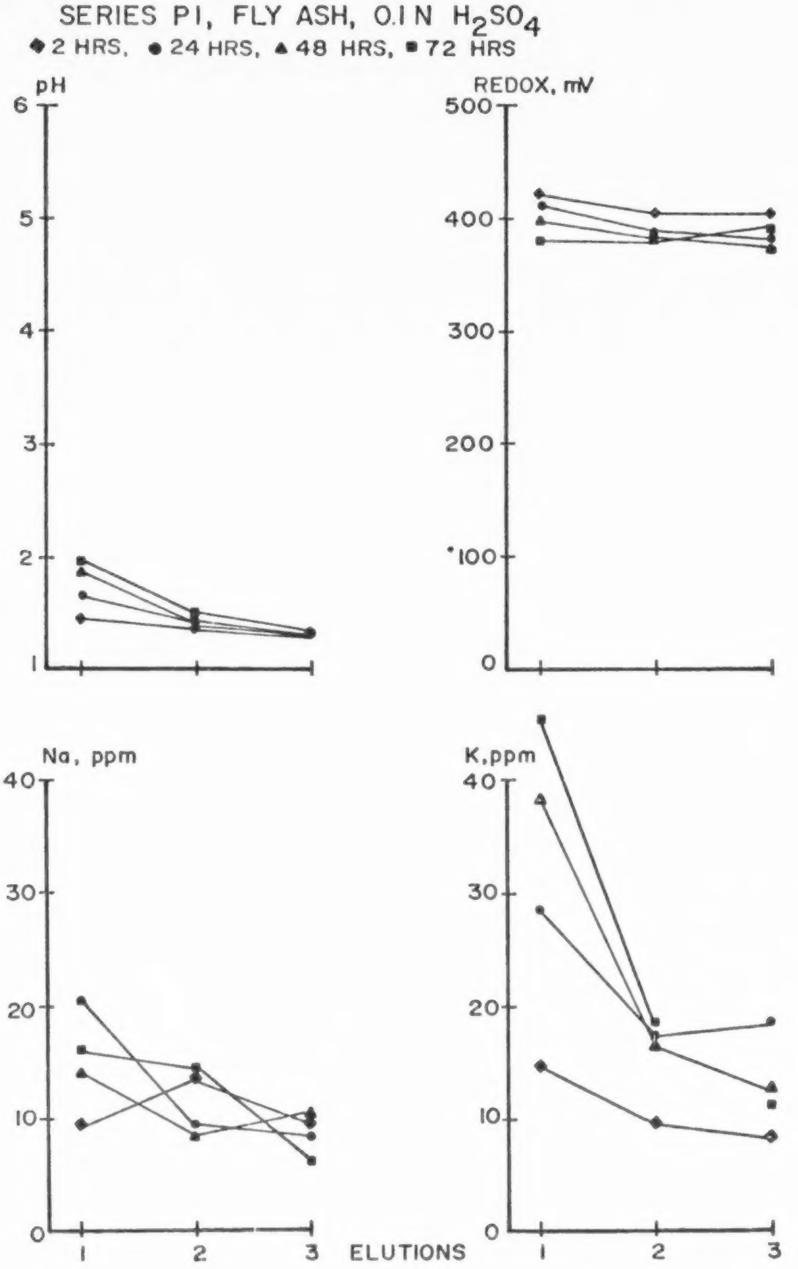

Figure A-15. Test P1 with fly ash and 0.1N H_2SO_4. pH, Redox, Na, K.

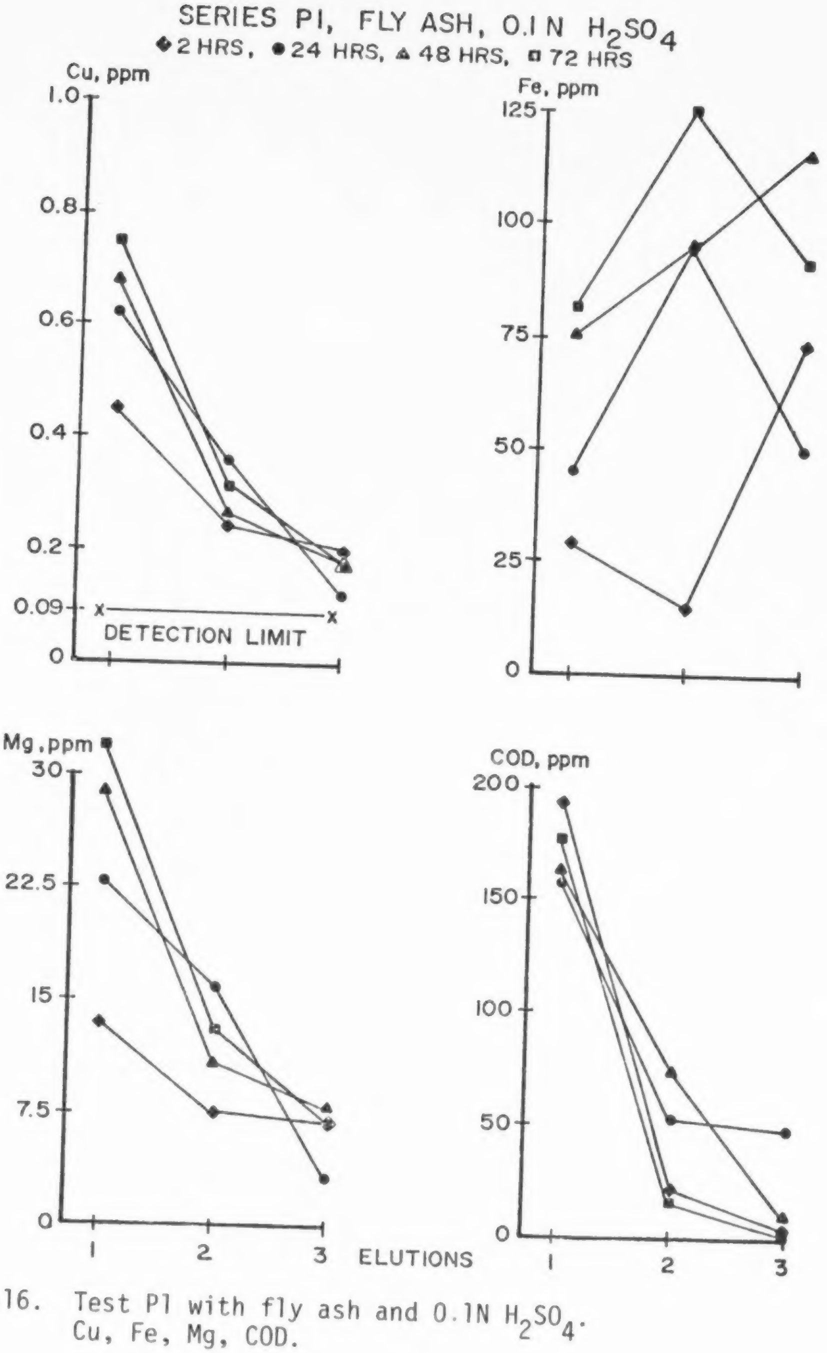

Figure A-16. Test P1 with fly ash and 0.1N H_2SO_4. Cu, Fe, Mg, COD.

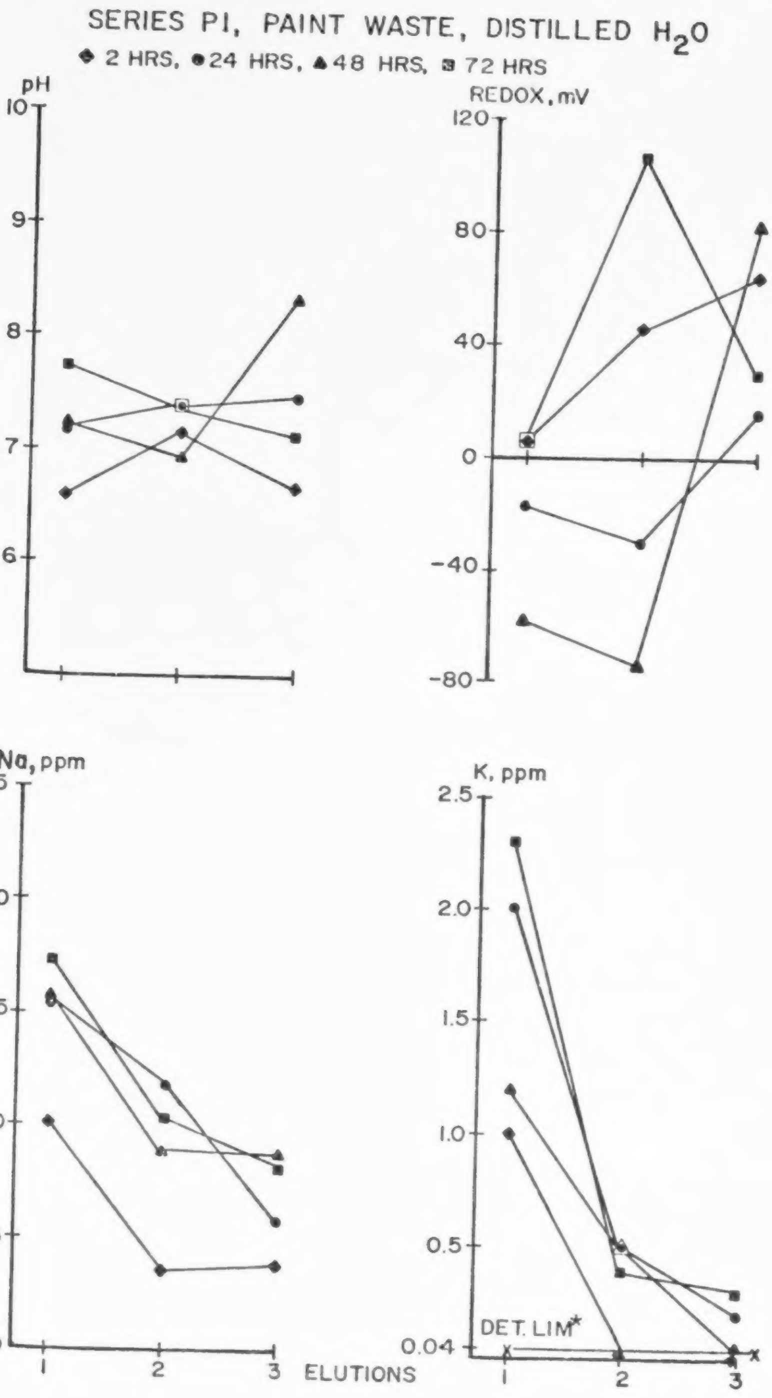

Figure A-17. Test P1 on the effect of time per elution using procedure R on paint waste with distilled water. pH, Redox, Na, K.

*Points below detection limit are approximate.

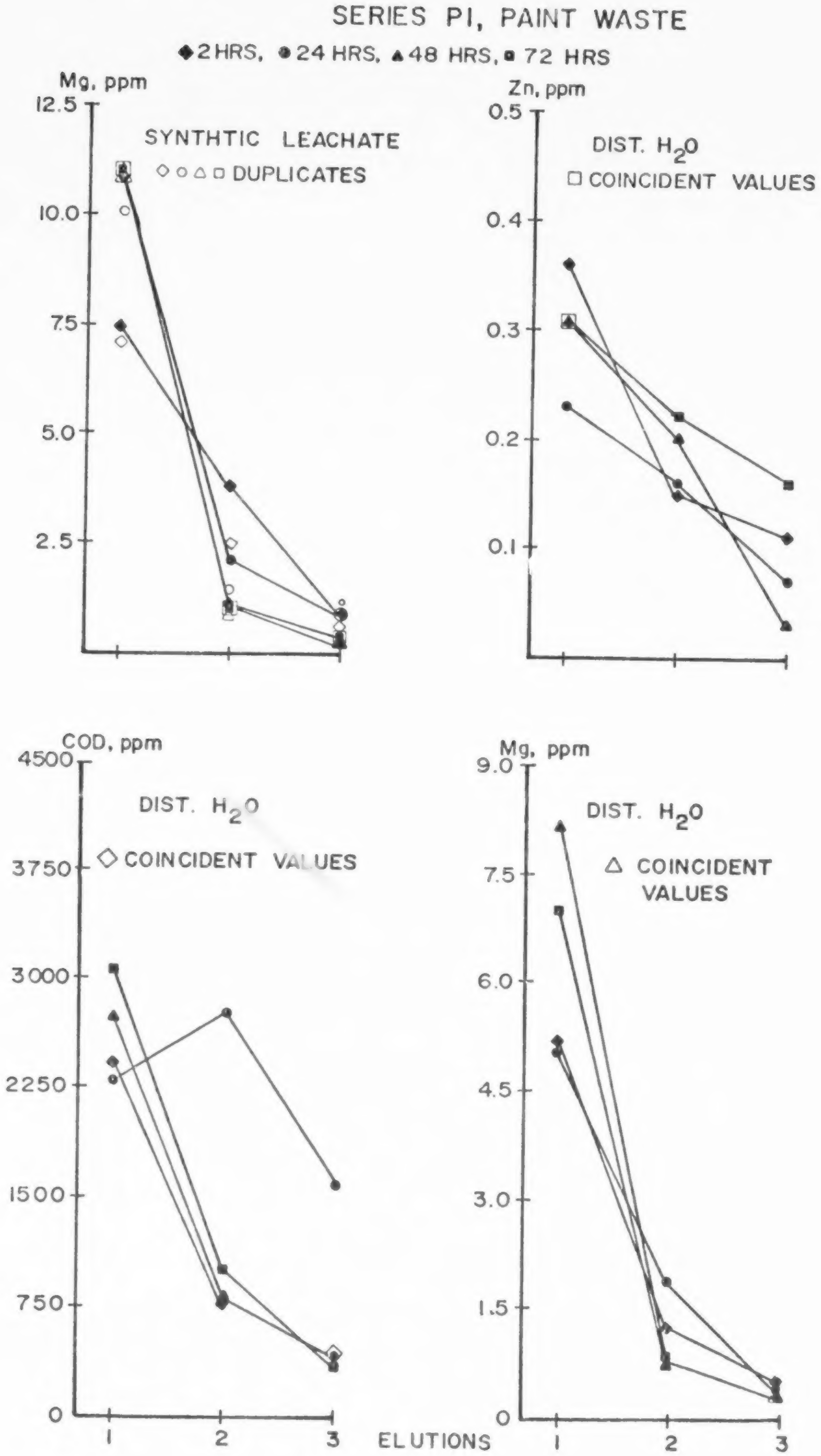

Figure A-18. Test P1 with paint waste and distilled water (Zn, COD, Mg) and synthetic leachate (Mg).

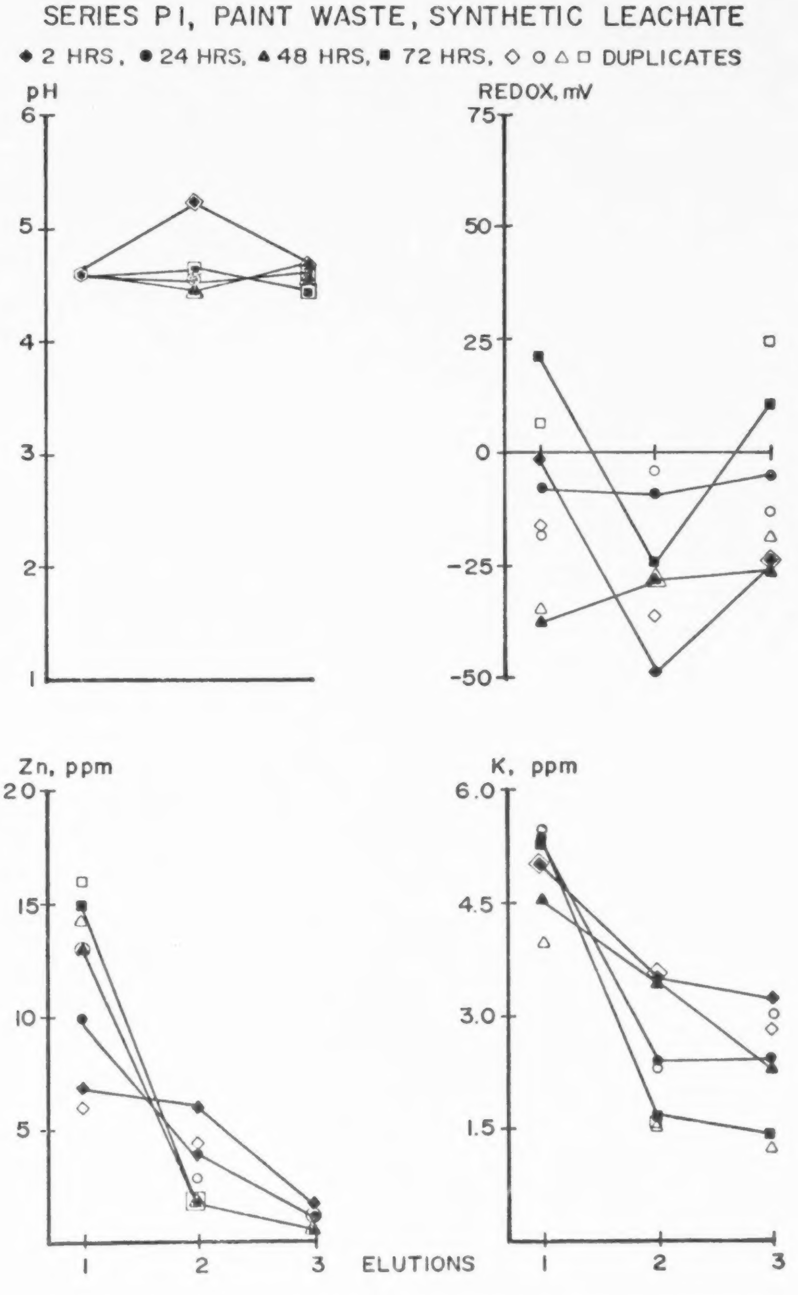

Figure A-19. Test P1 with paint waste and synthetic leachate. pH, Redox, Zn, and K.

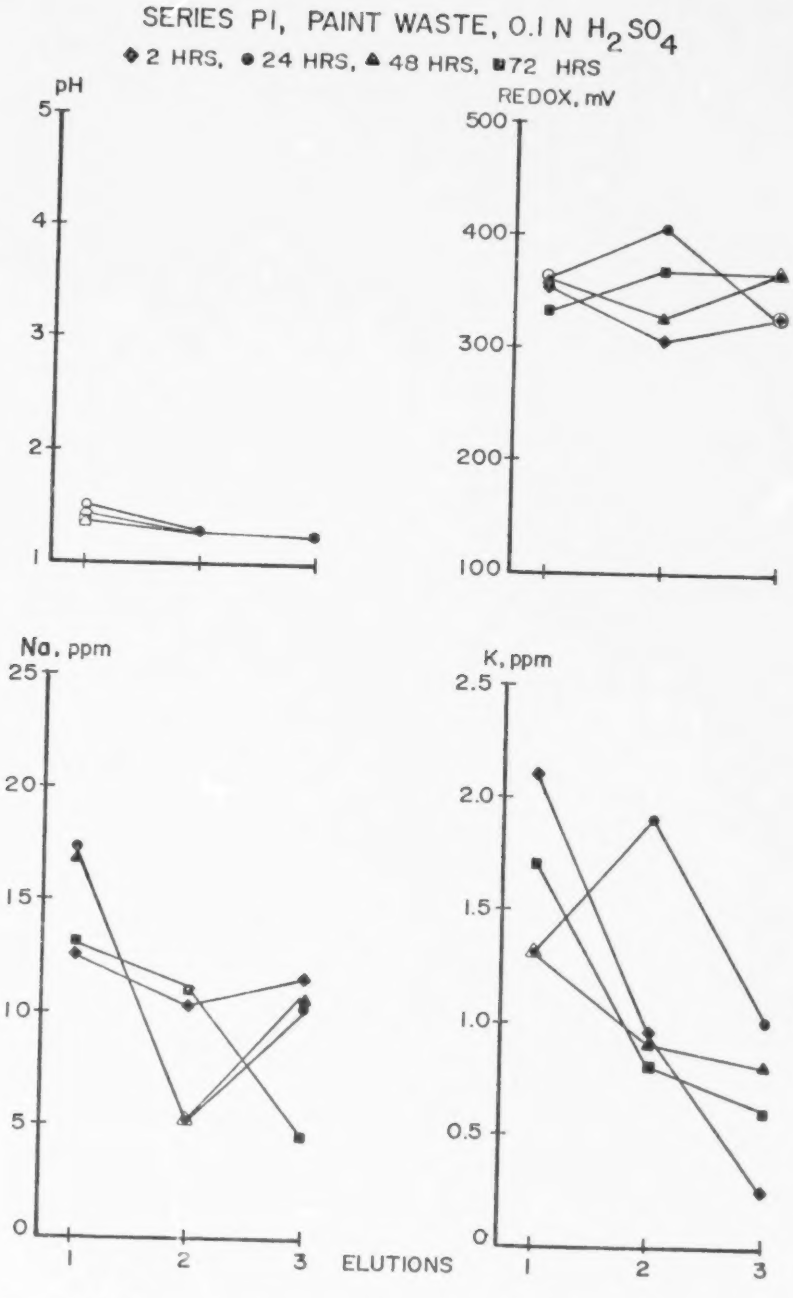

Figure A-20. Test P1 with paint waste and 0.1N H_2SO_4. pH, Redox, Na, K.

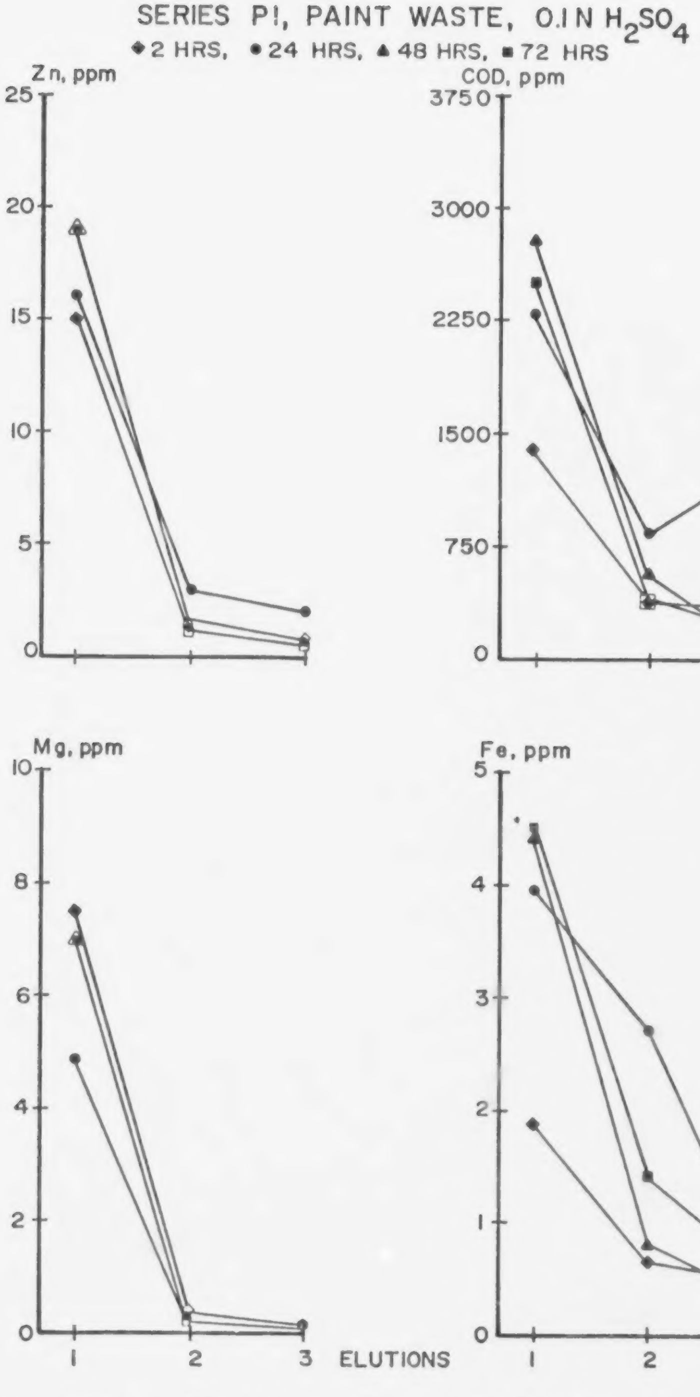

Figure A-21. Test P1 with paint waste and 0.1N H_2SO_4. Zn, COD, Mg, Fe.

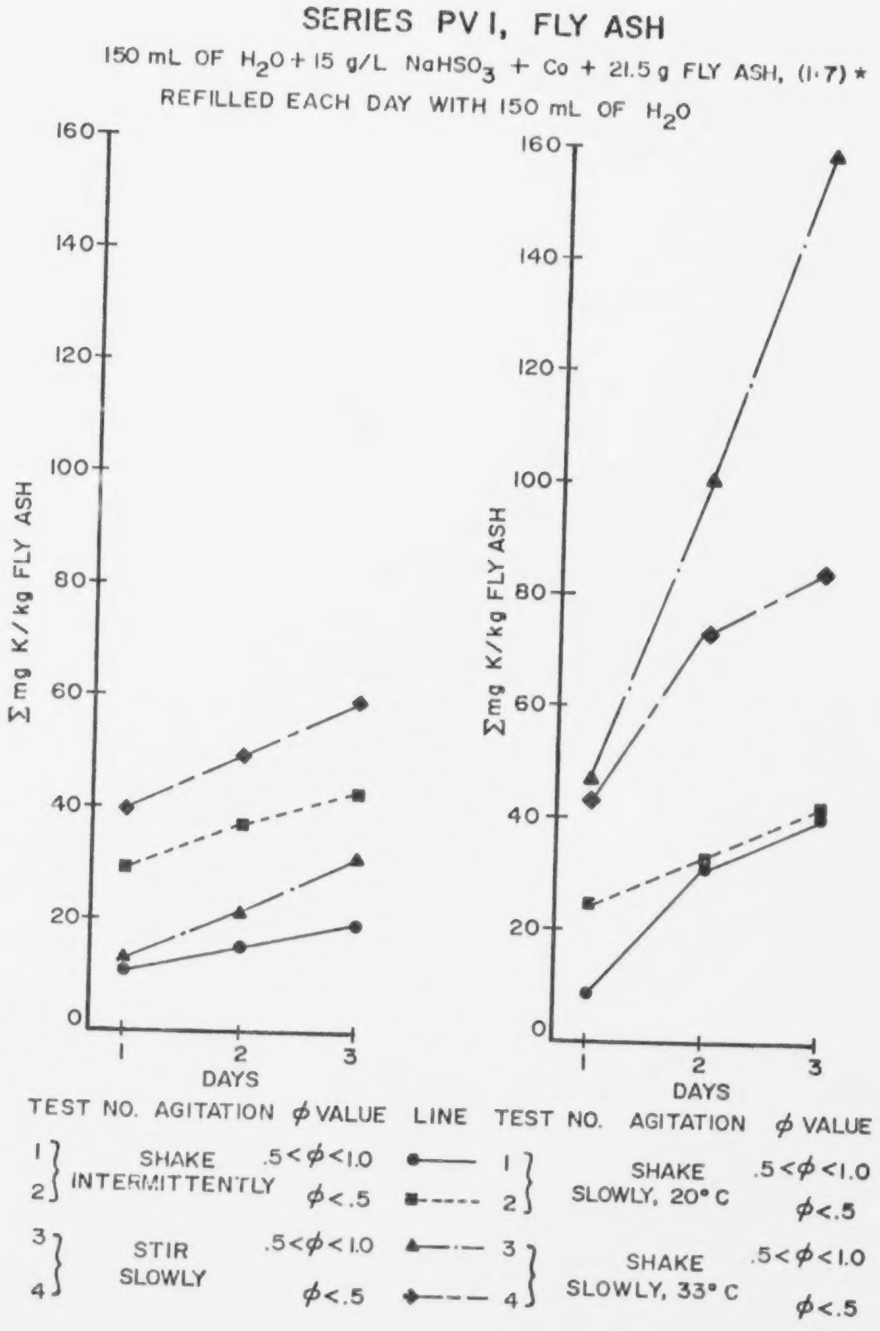

Figure A-22. Test PV1 using different agitation techniques on fly ash with distilled water. (See text for procedure.) K.

*$NaHSO_3$ and Co (trace) are to remove dissolved oxygen in the leach mixture.

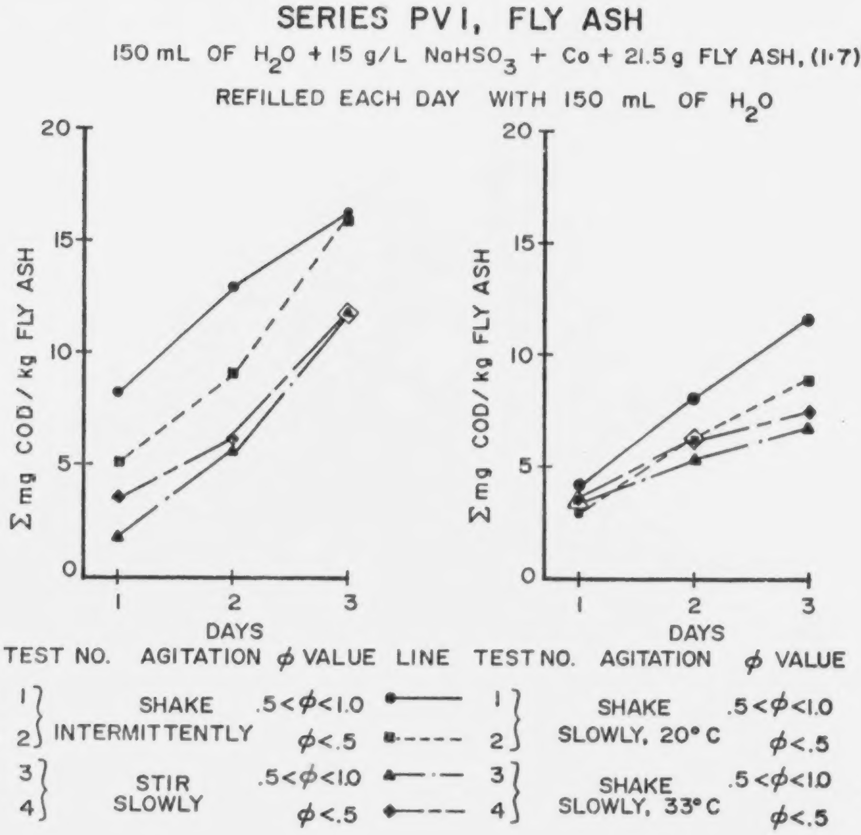

Figure A-23. Test PV1 using different agitation techniques on fly ash with distilled water. COD.

*See Figure A-22 footnote.

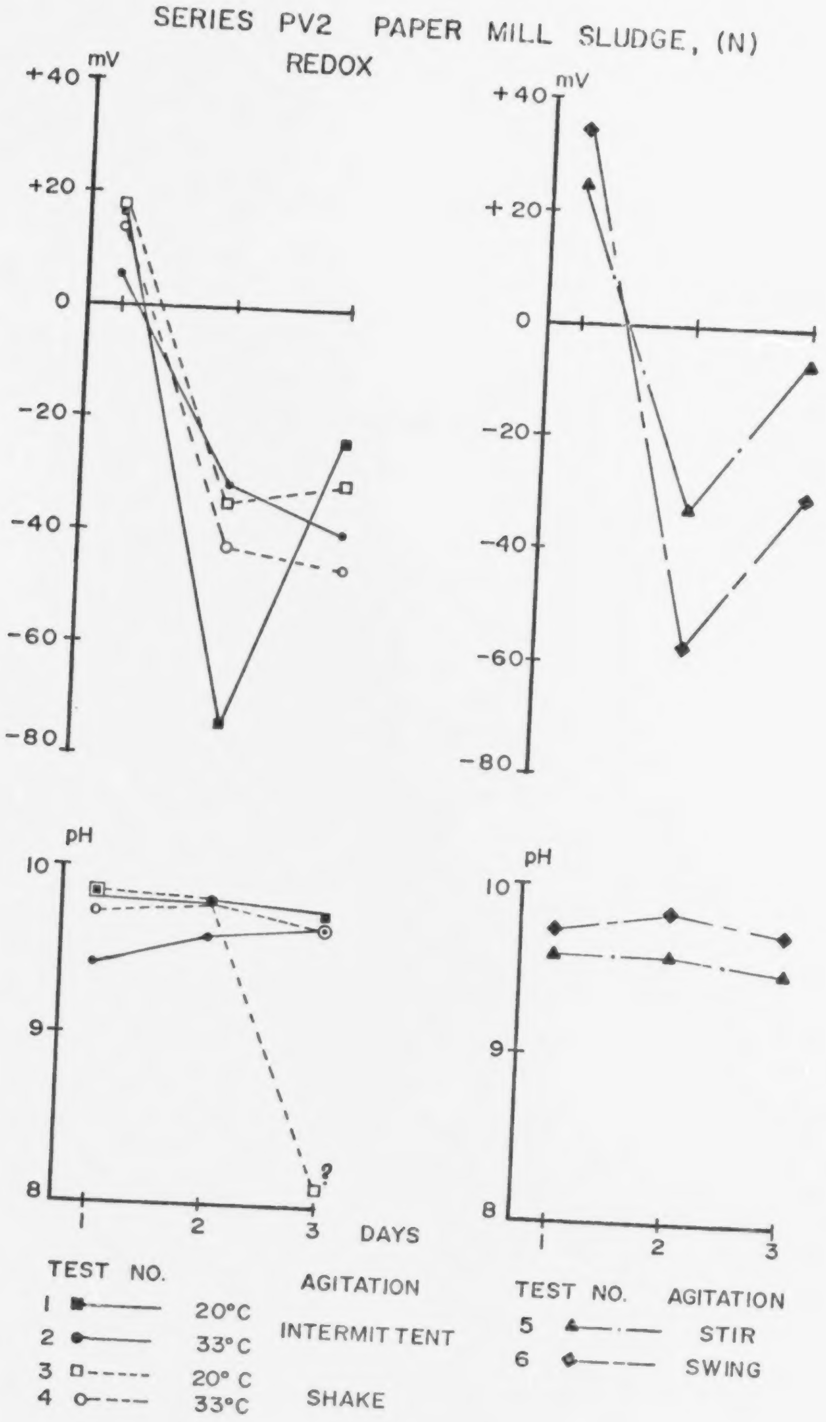

Figure A-24. Test PV2 using different agitation techniques on papermill sludge with distilled water. (See test for procedure.) Redox, pH.

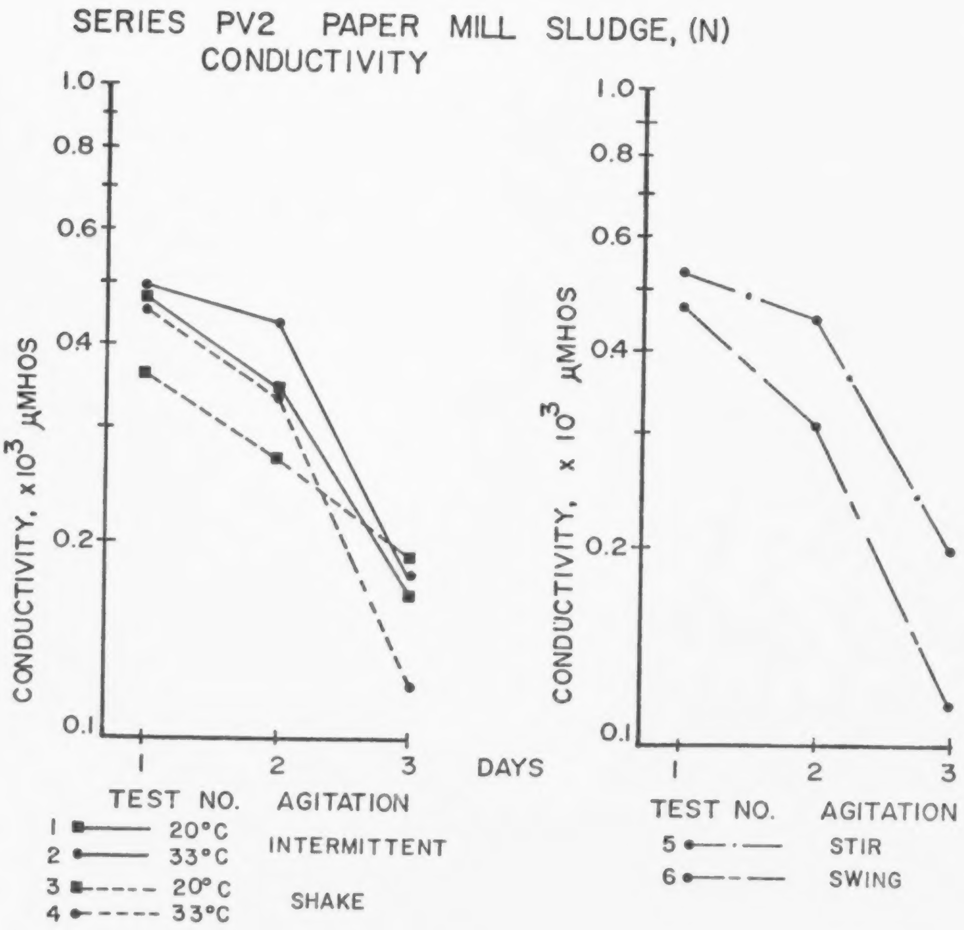

Figure A-25. Test PV2 using different agitation techniques on papermill sludge with distilled water. Specific conductance.

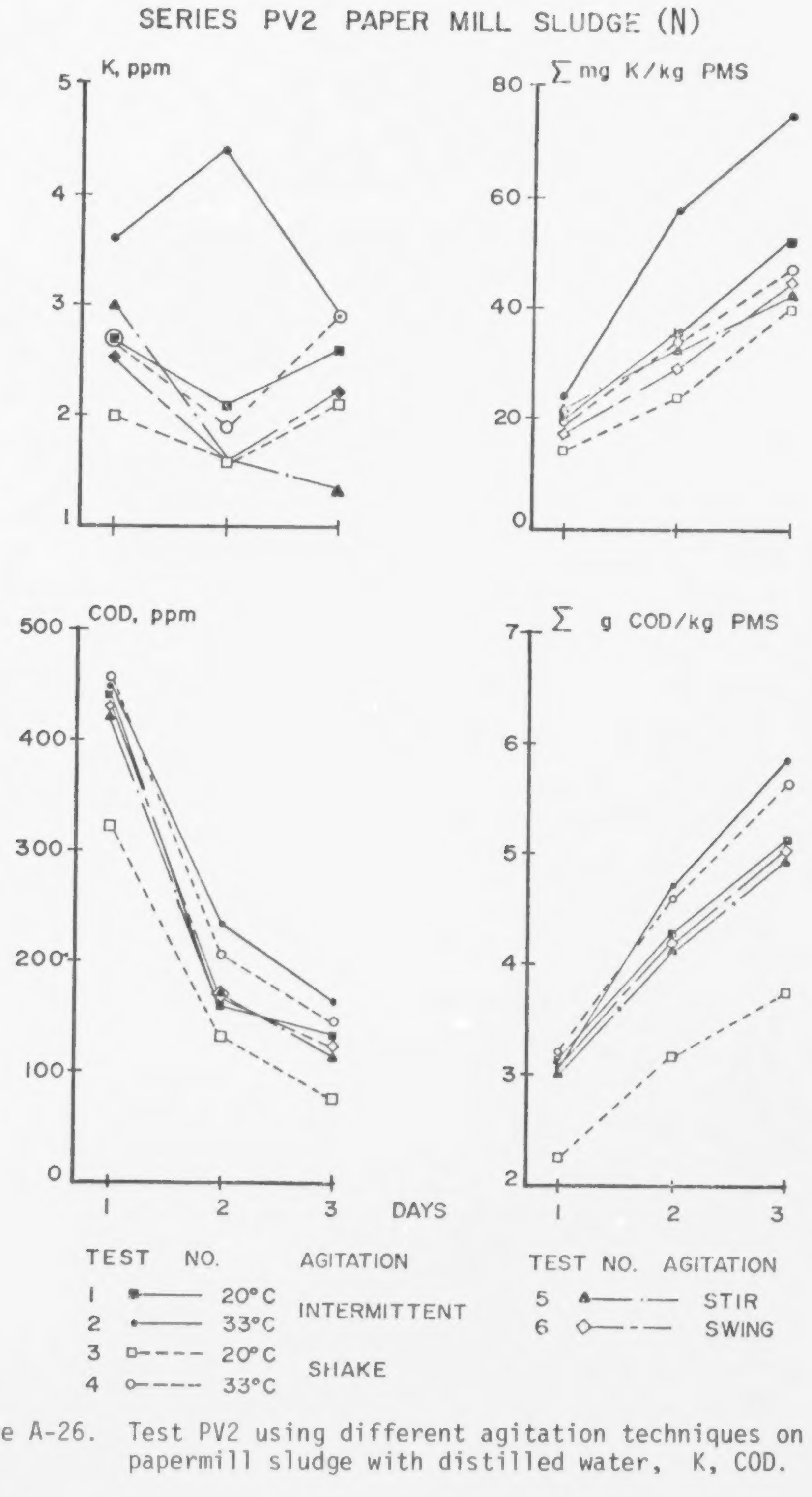

Figure A-26. Test PV2 using different agitation techniques on papermill sludge with distilled water, K, COD.

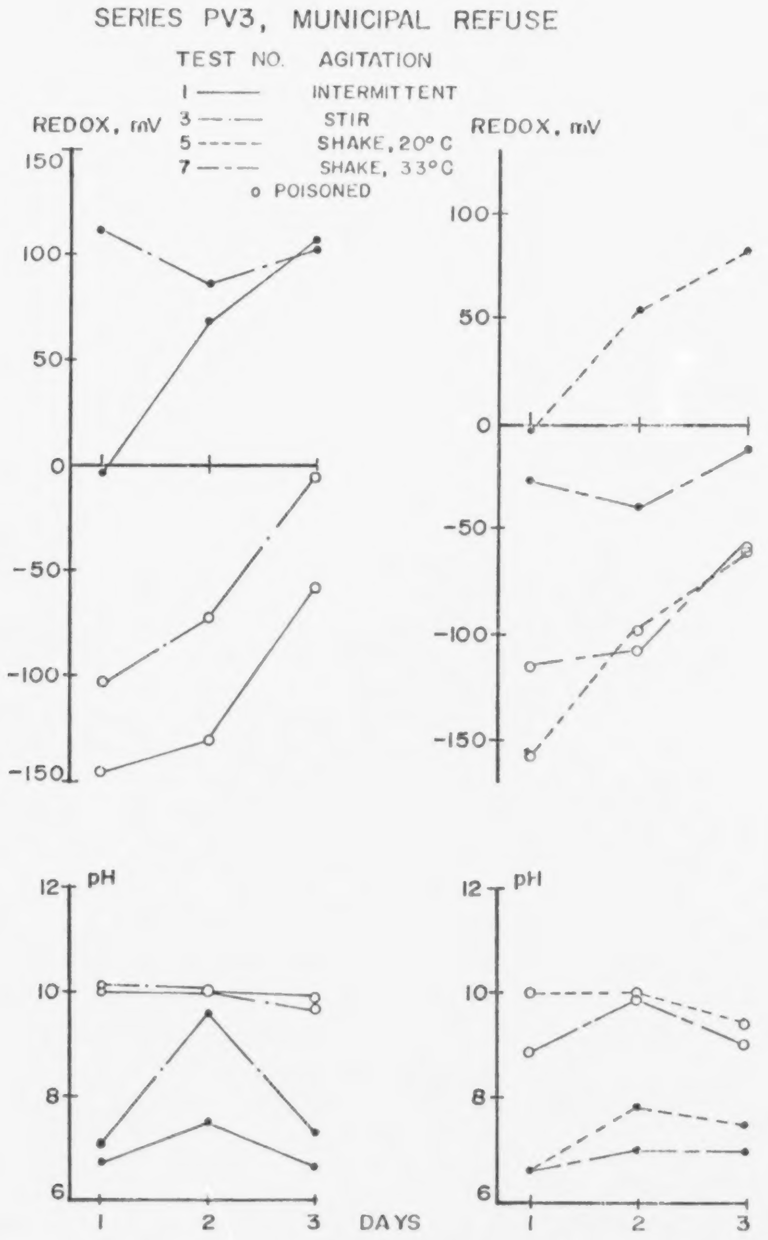

Figure A-27. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. (See text for procedure.) Redox, pH.

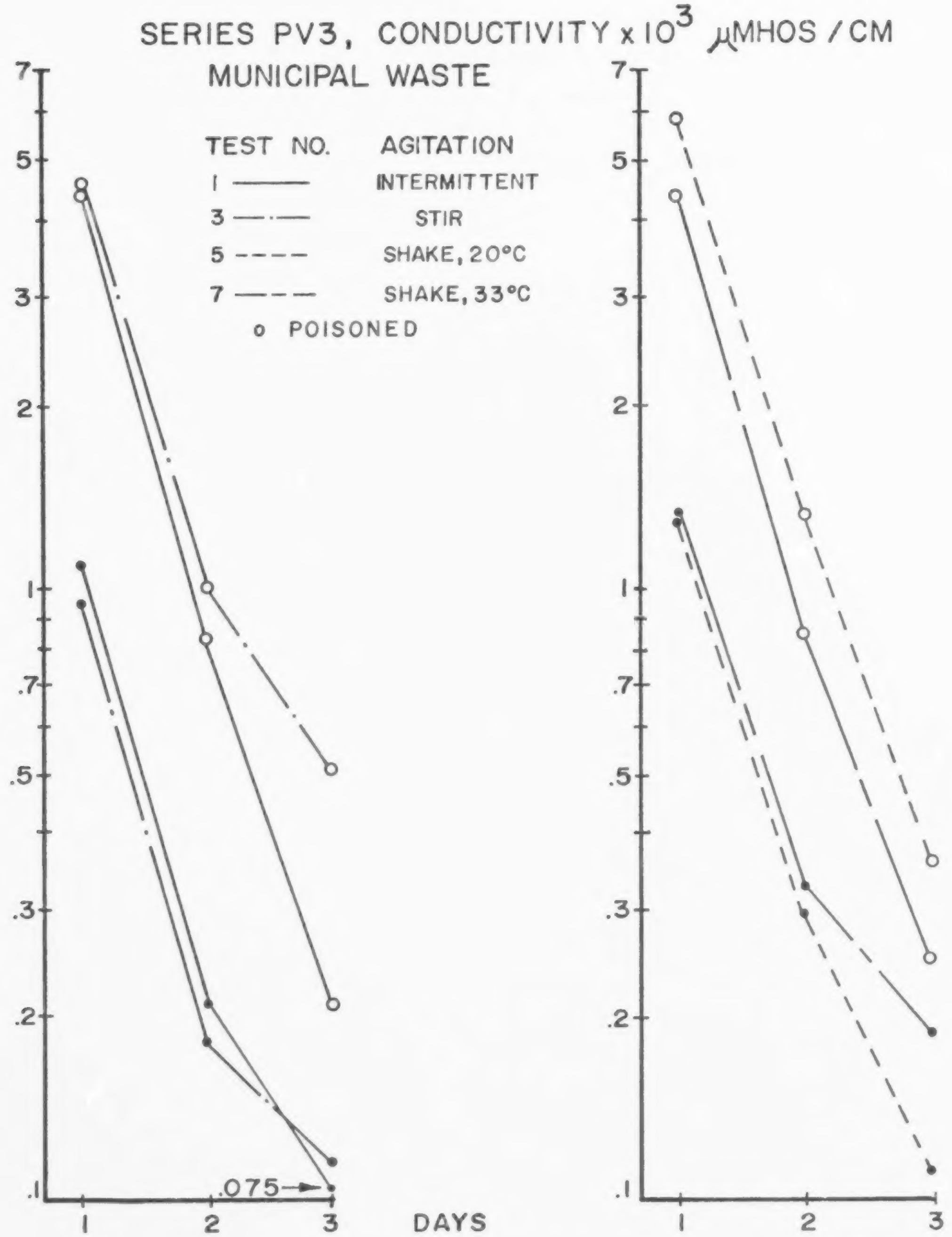

Figure A-28. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. Specific conductance.

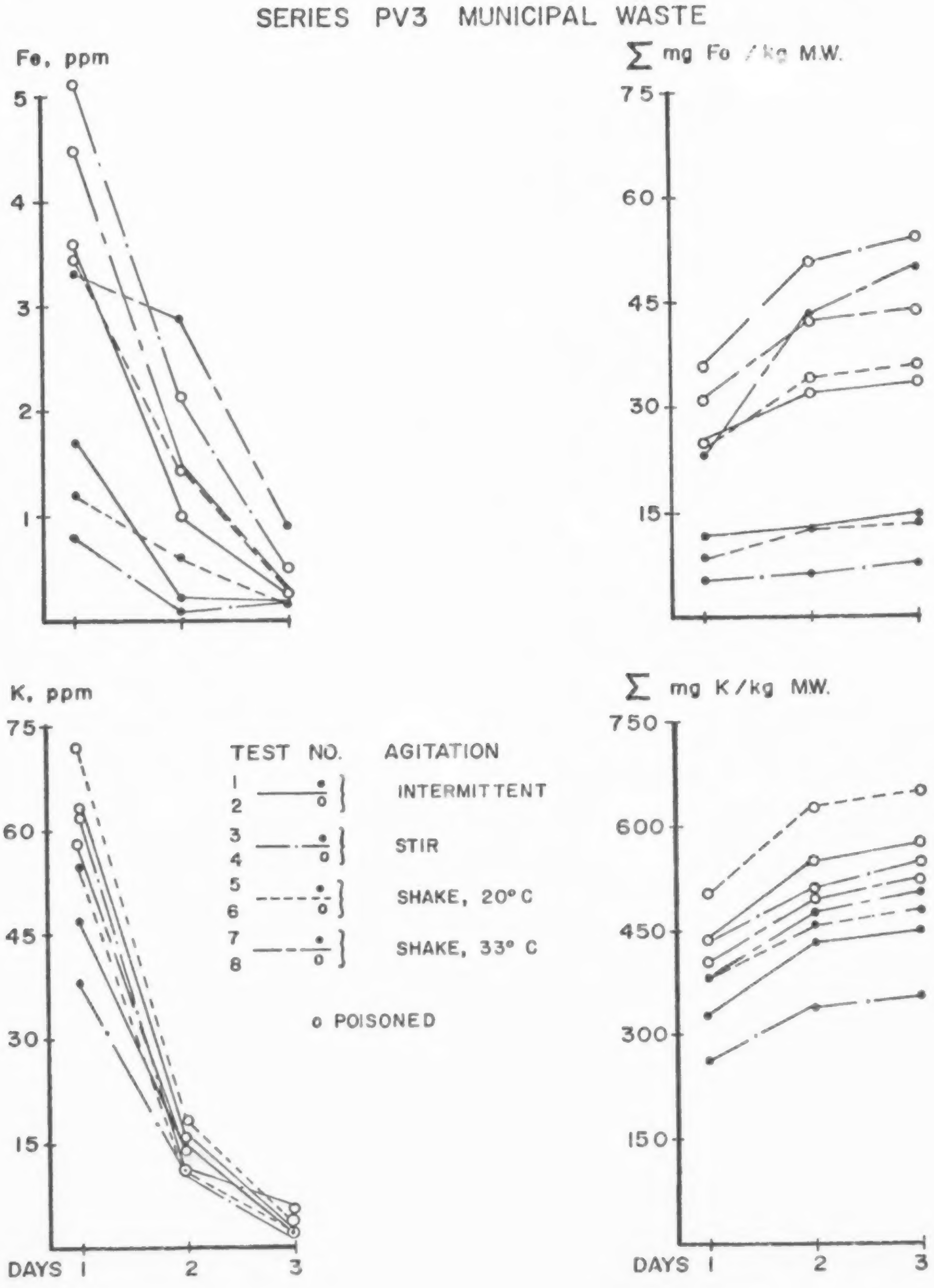

Figure A-29. Test PV 3 using different agitation techniques on shredded municipal refuse with distilled water. Fe, K.

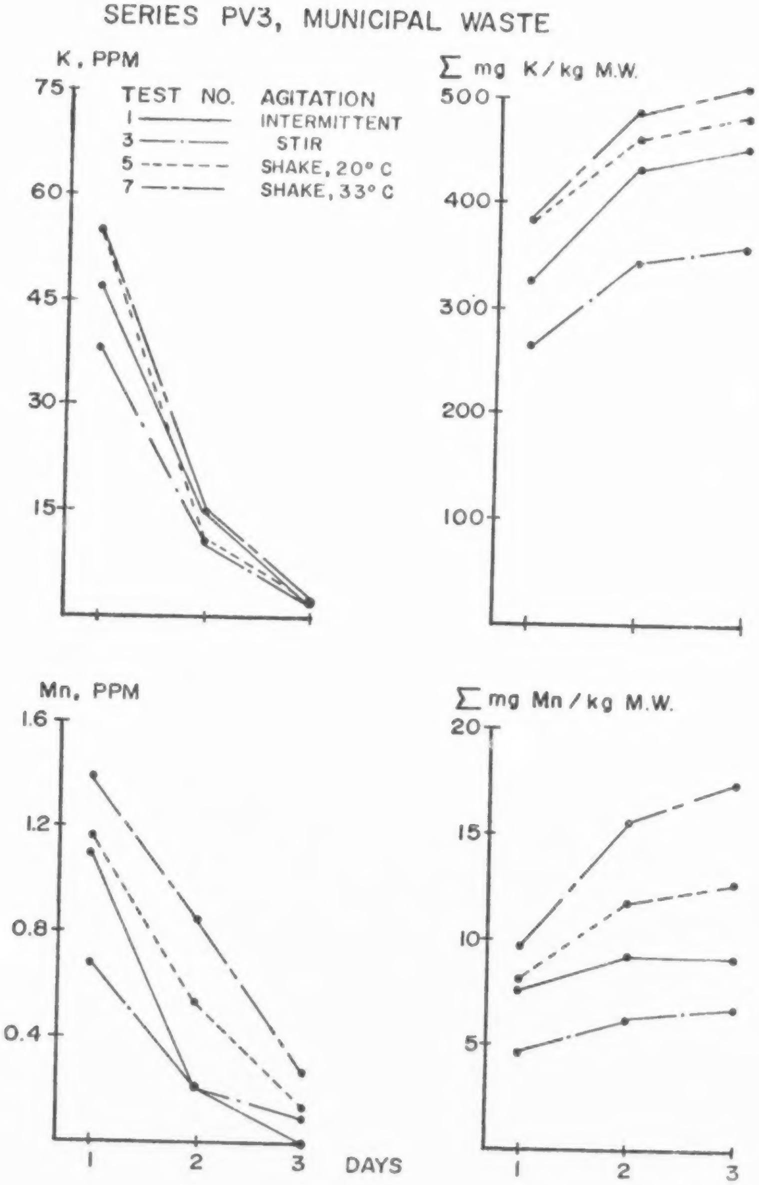

Figure A-30. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. K (repeat), Mn.

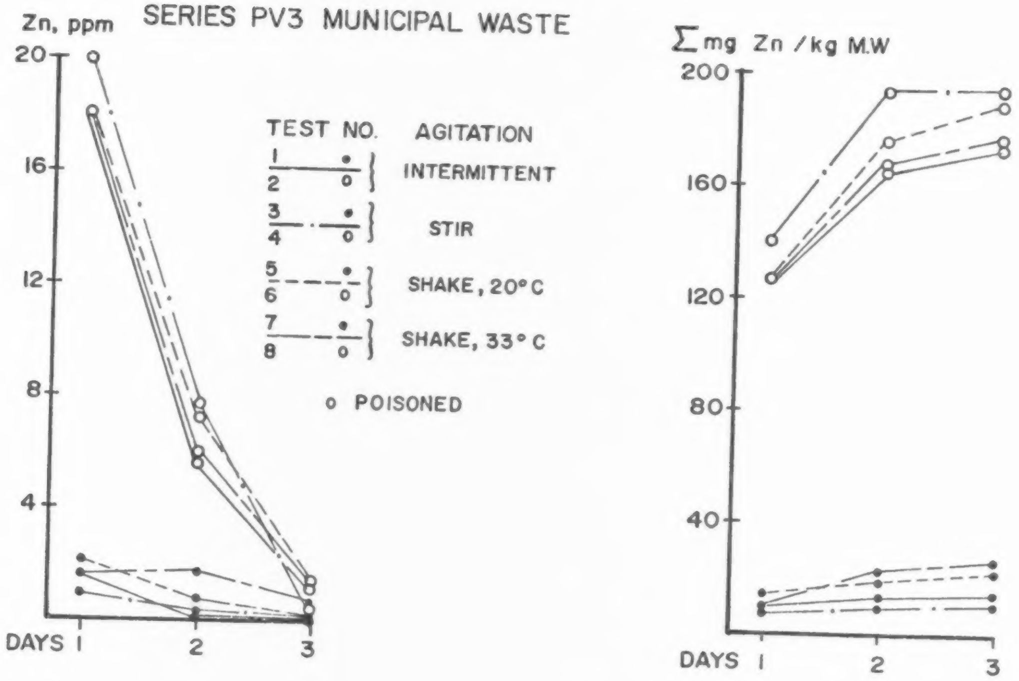

Figure A-31. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. Zn.

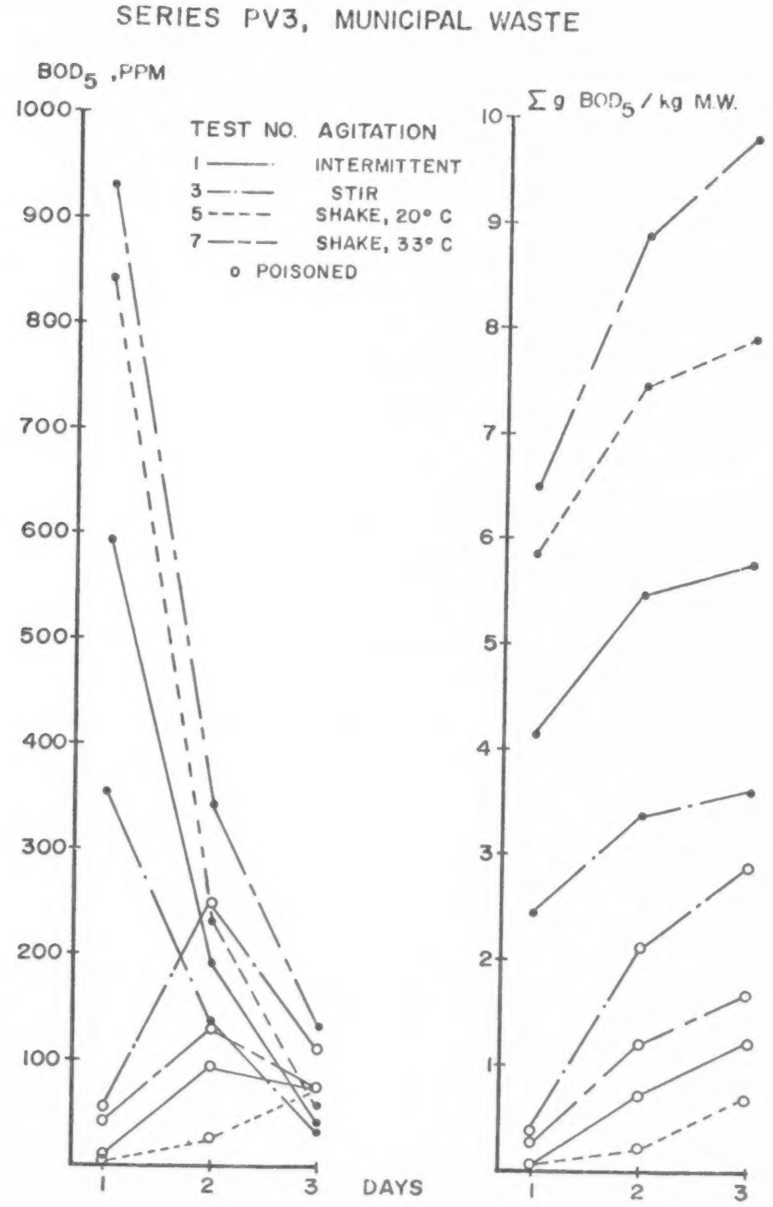

Figure A-32. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. BOD.

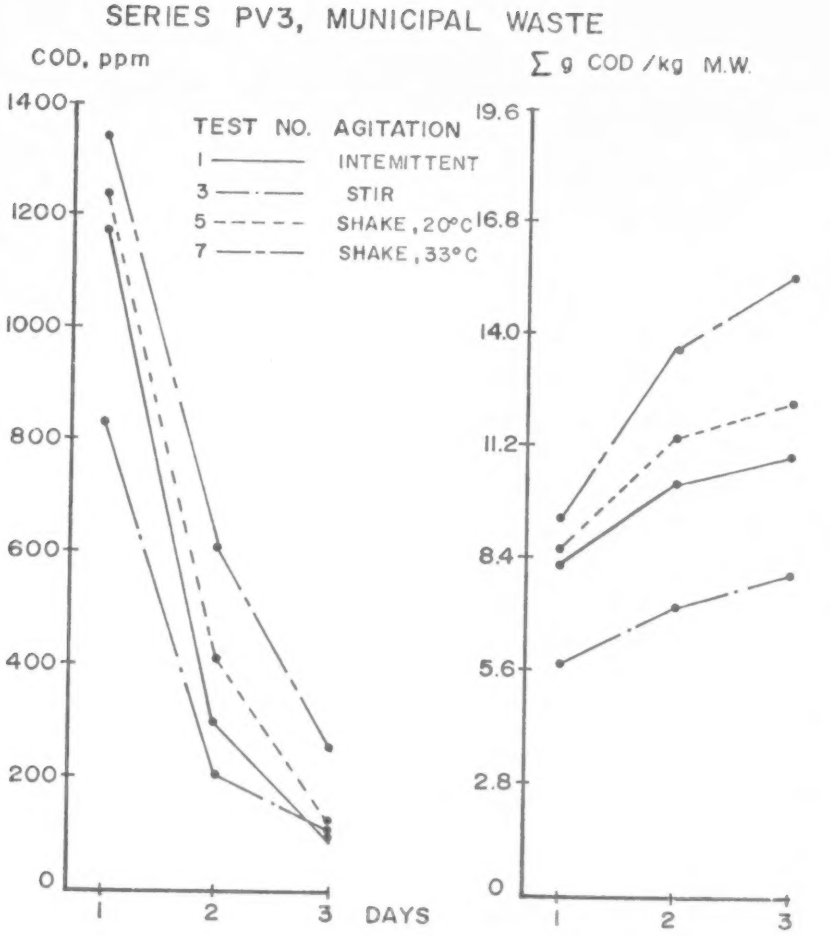

Figure A-33. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. COD.

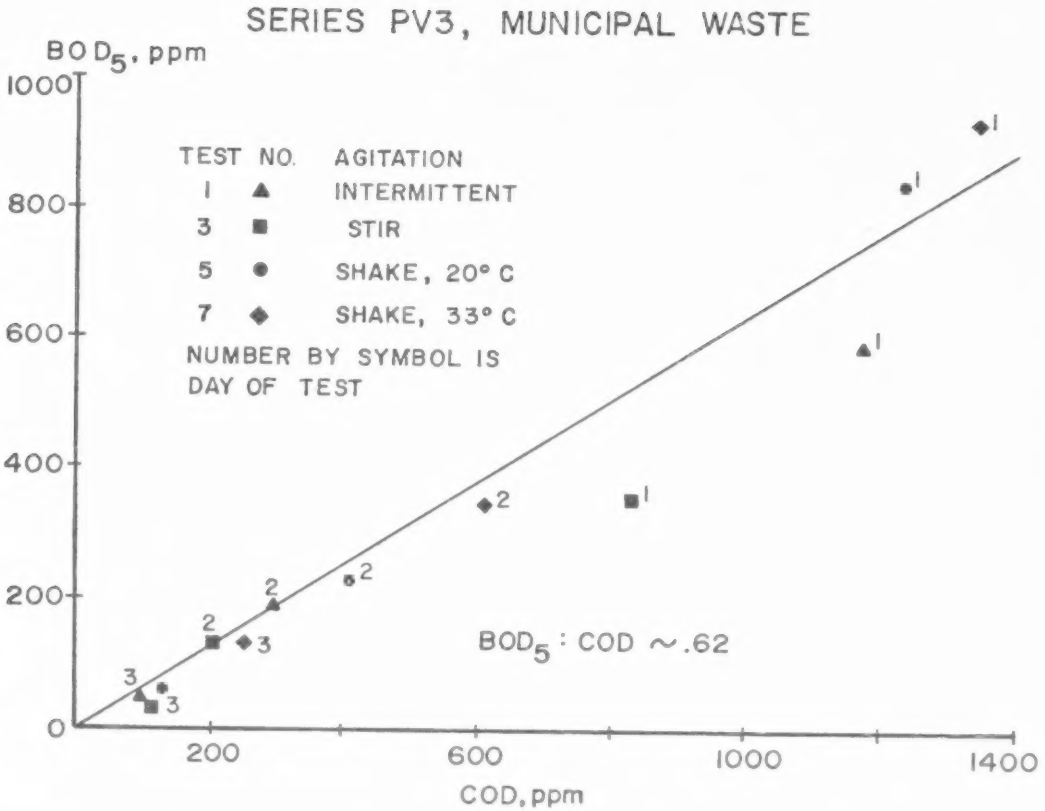

Figure A-34. Test PV3 using different agitation techniques on shredded municipal refuse with distilled water. Comparison of BOD and COD concentrations.

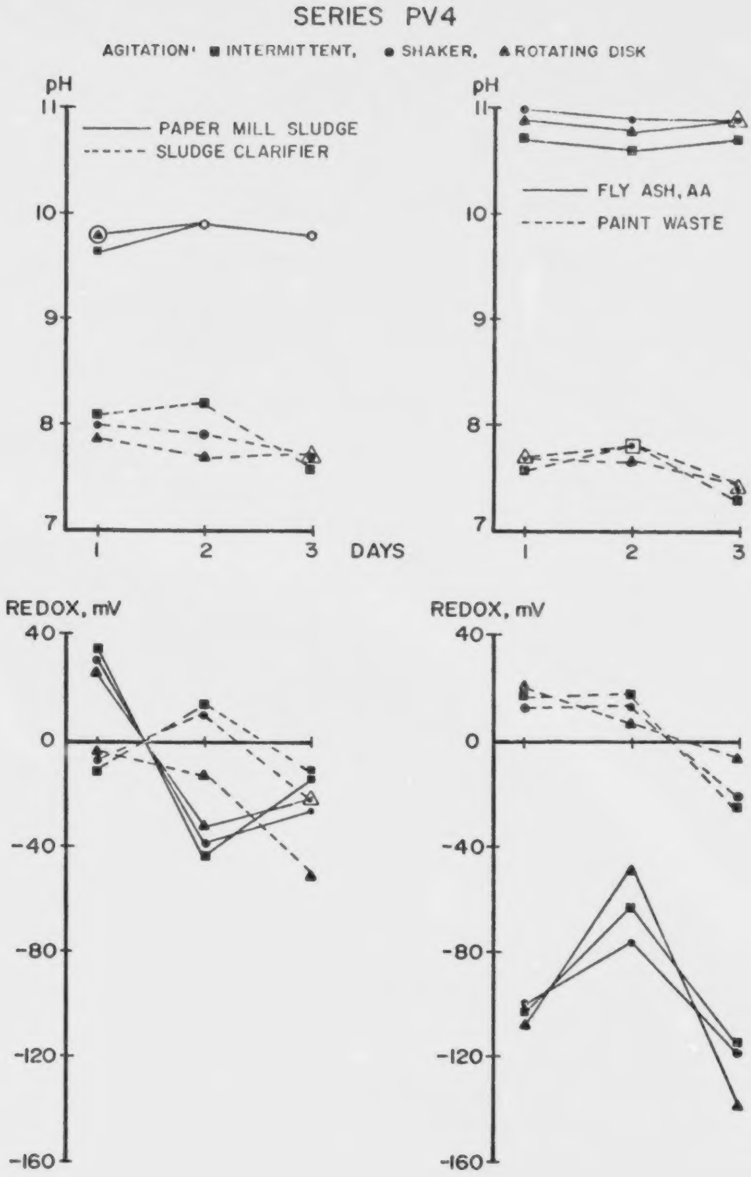

Figure A-35. Test PV4 comparing different agitation techniques on four wastes with distilled water. (See text for procedure.) pH and Redox.

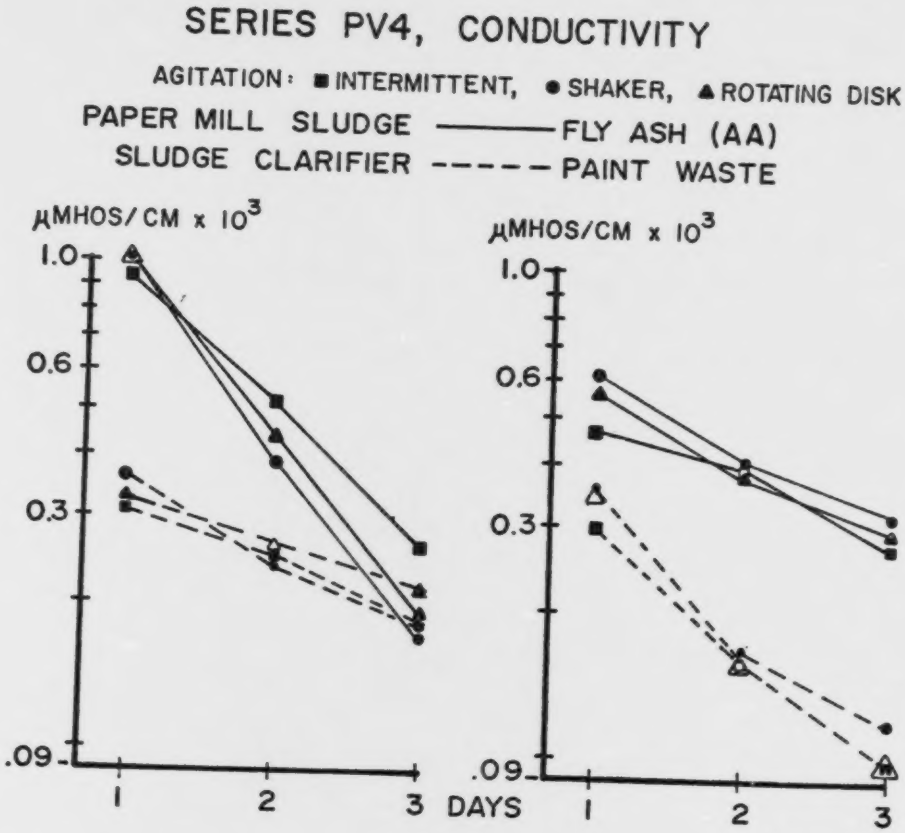

Figure A-36. Test PV4 comparing different agitation techniques on four wastes with distilled water. Specific conductance.

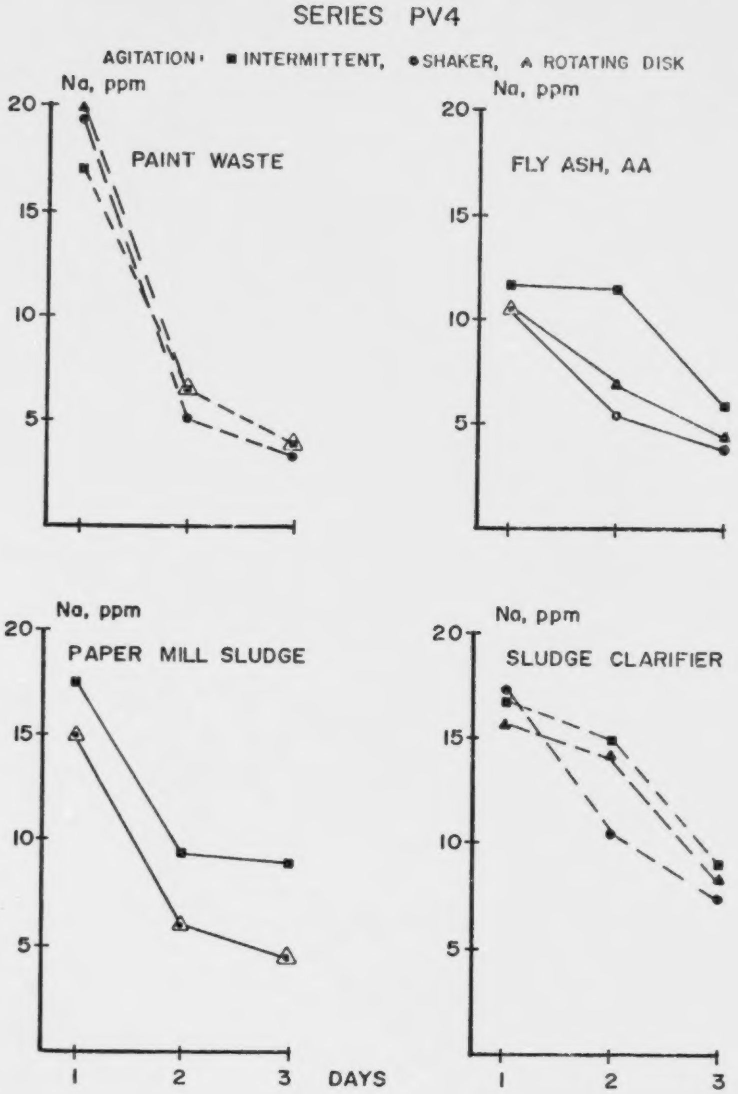

Figure A-37. Test PV4 comparing different agitation techniques on four wastes with distilled water. Na.

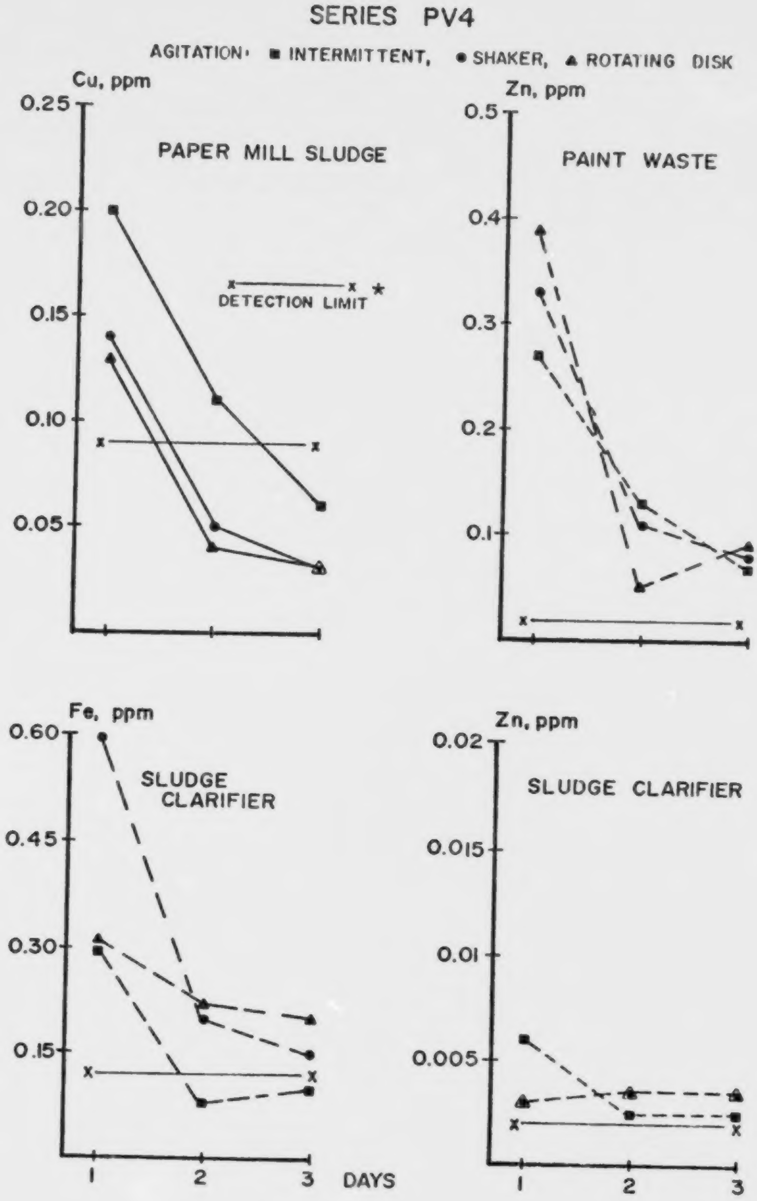

Figure A-38. Test PV4 comparing different agitation techniques on four wastes with distilled water. Cu, Zn, Fe.

*All points below detection limit are approximate.

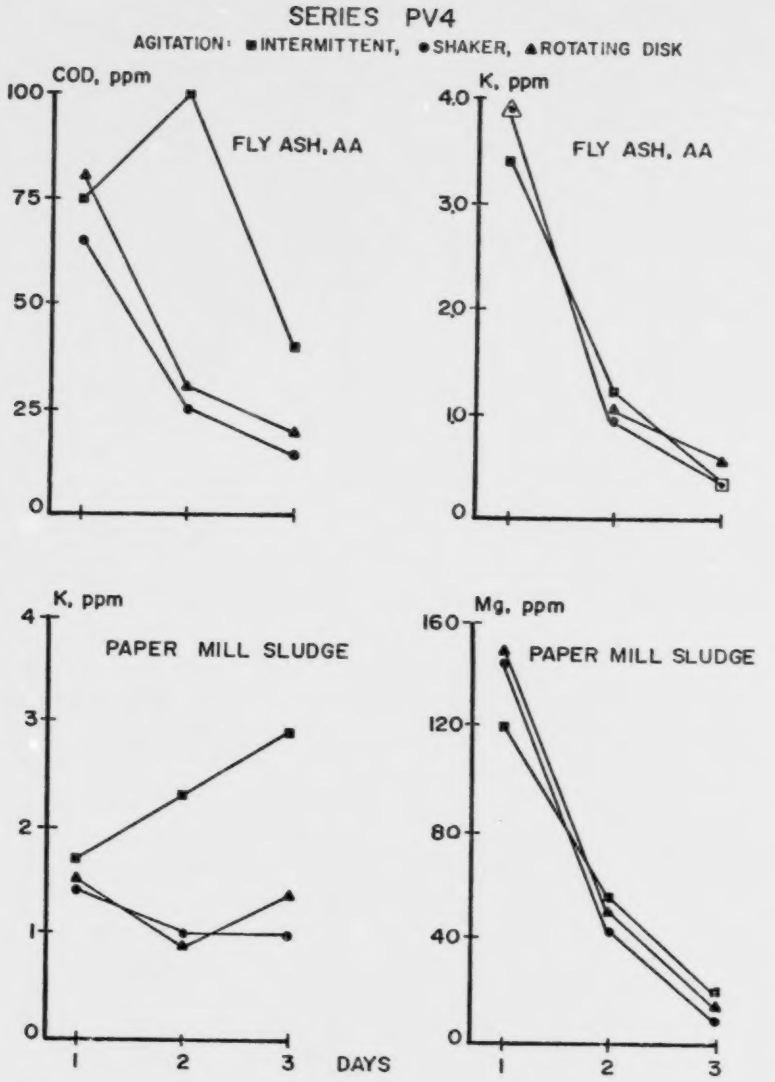

Figure A-39. Test PV4 comparing different agitation techniques on four wastes with distilled water. COD, K, Mg.

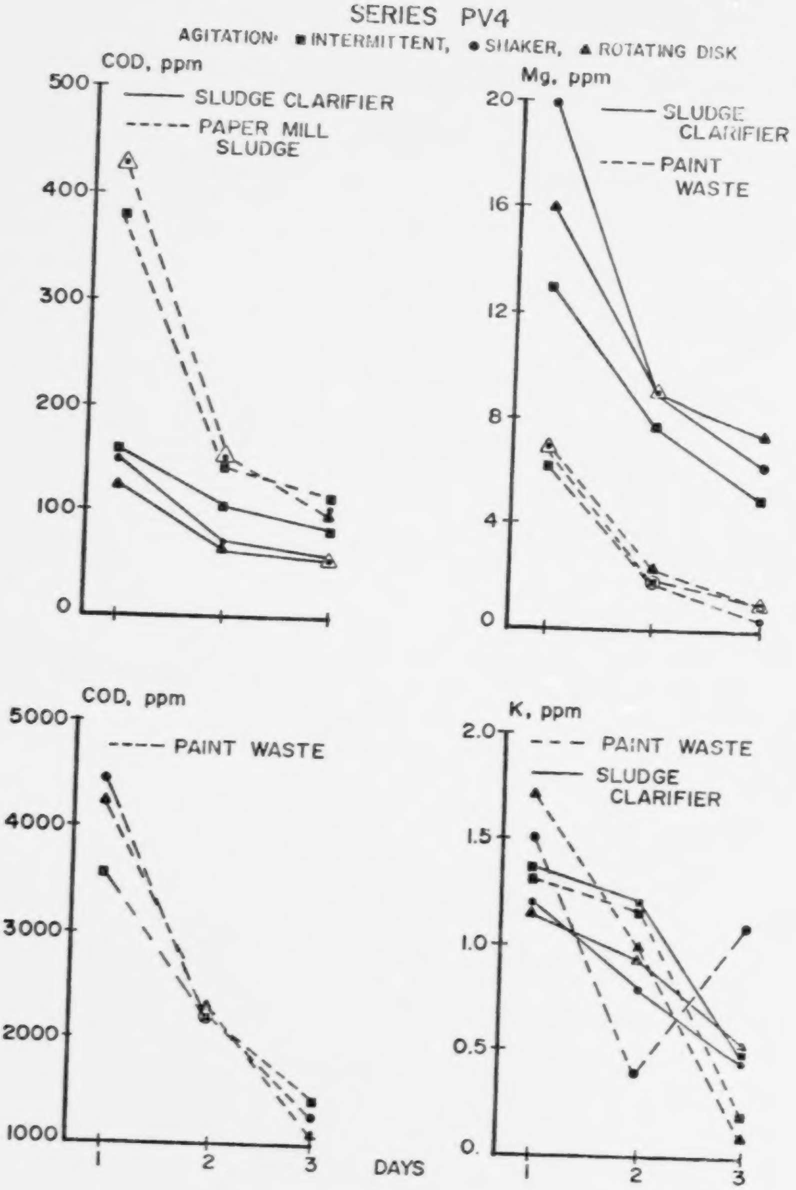

Figure A-40. Test PV4 comparing different agitation techniques on four wastes with distilled water. COD, Mg, K.

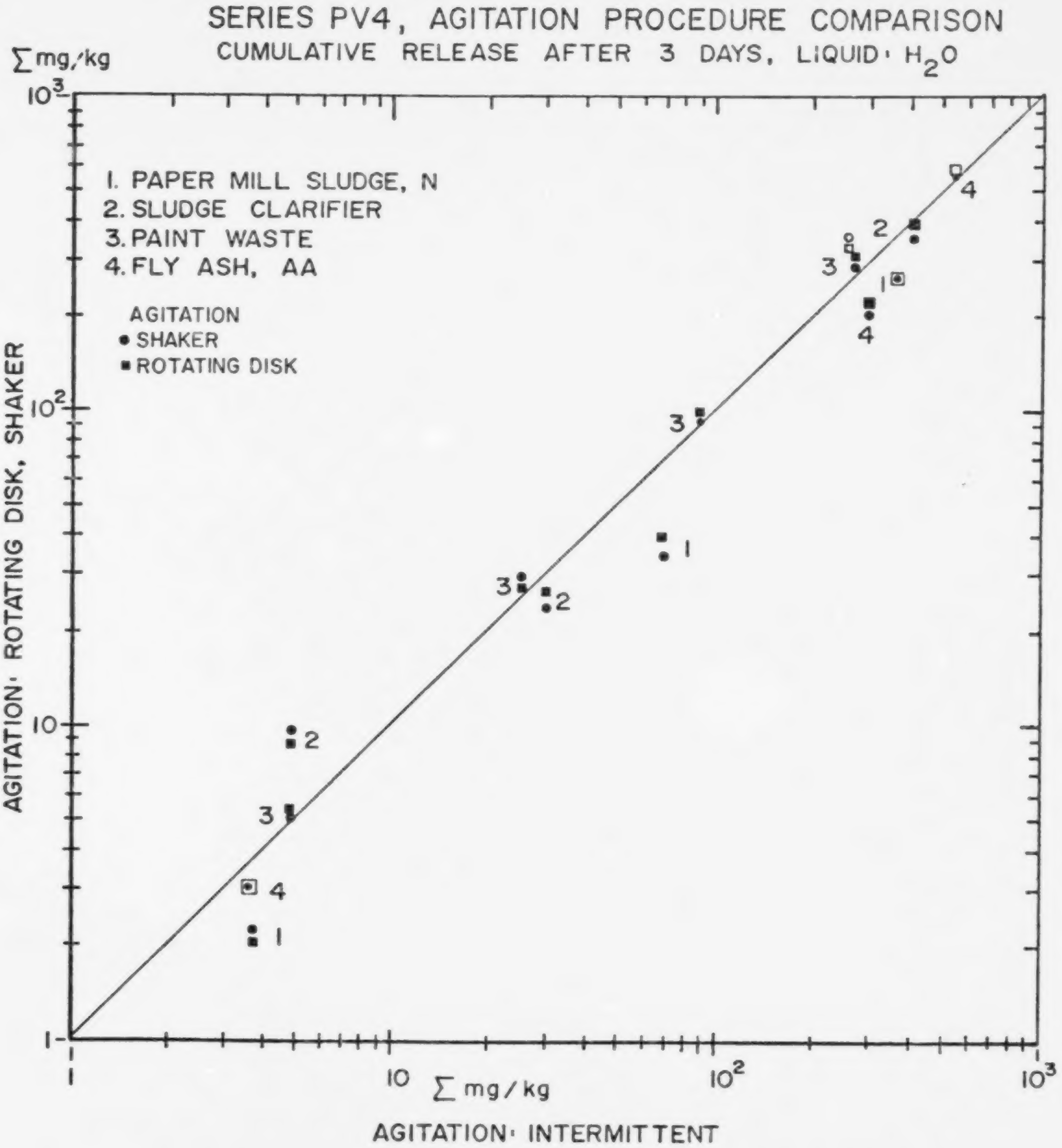

Figure A-41. Test PV4 comparing the cumulative release of all measured parameters after 3 elutions using the rotating disc and intermittent shaking agitation techniques.

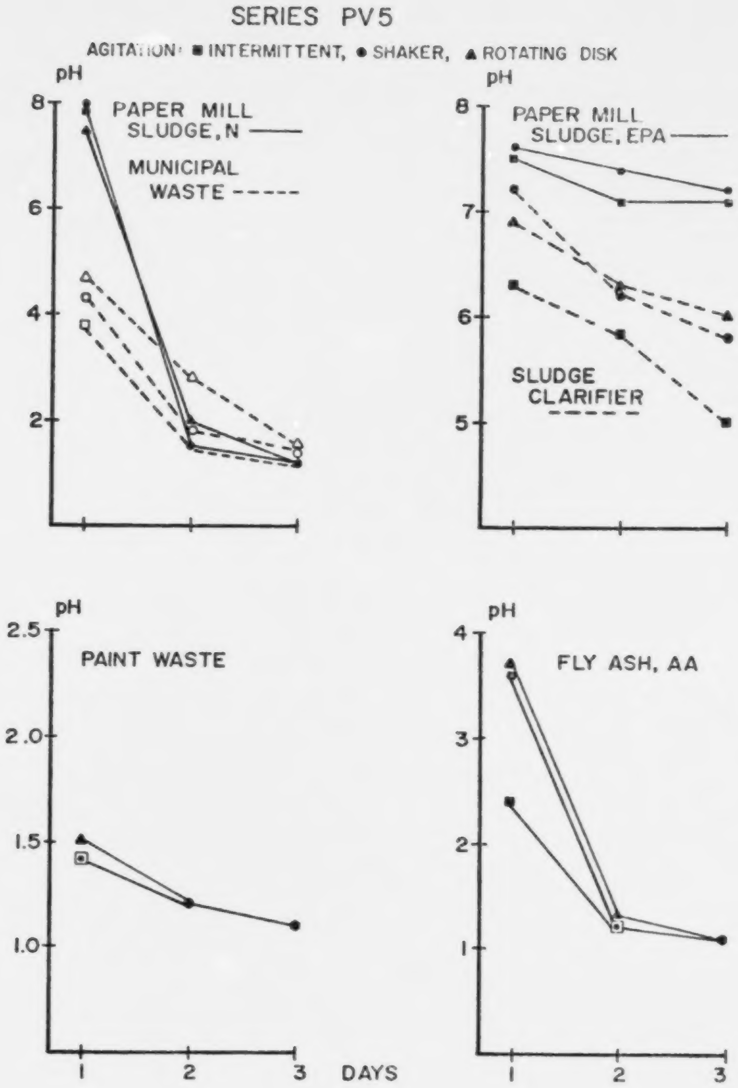

Figure A-42. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. (See text for procedure.) pH.

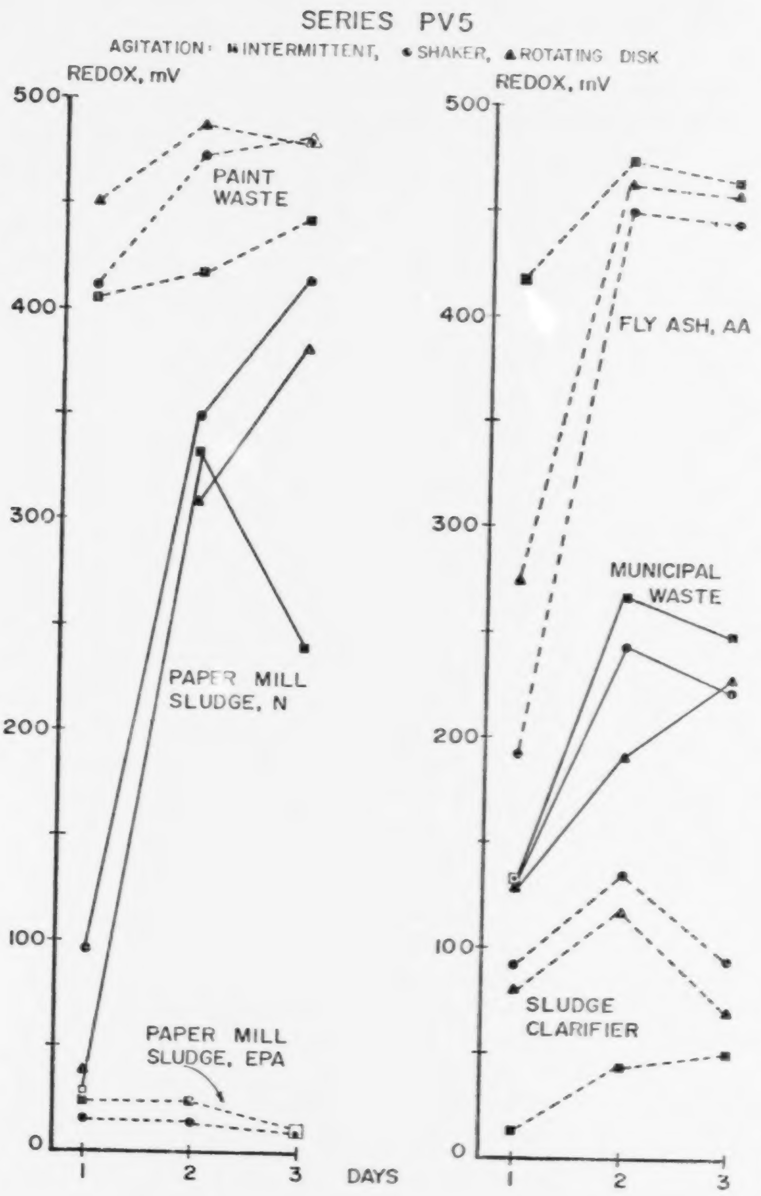

Figure A-43. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Redox.

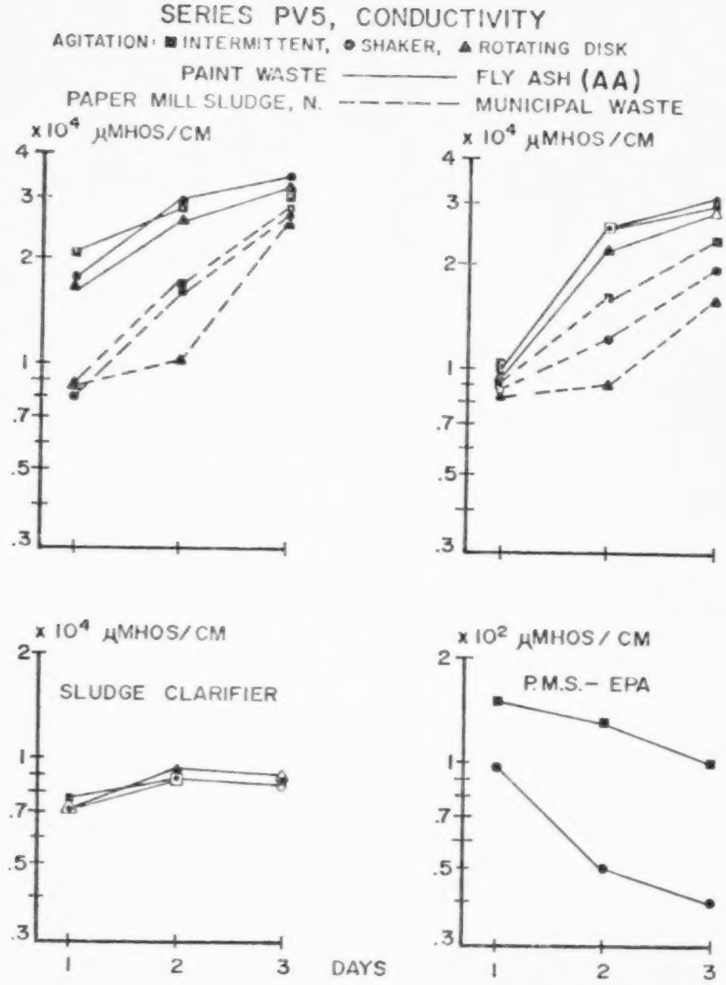

Figure A-44. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Specific conductance.

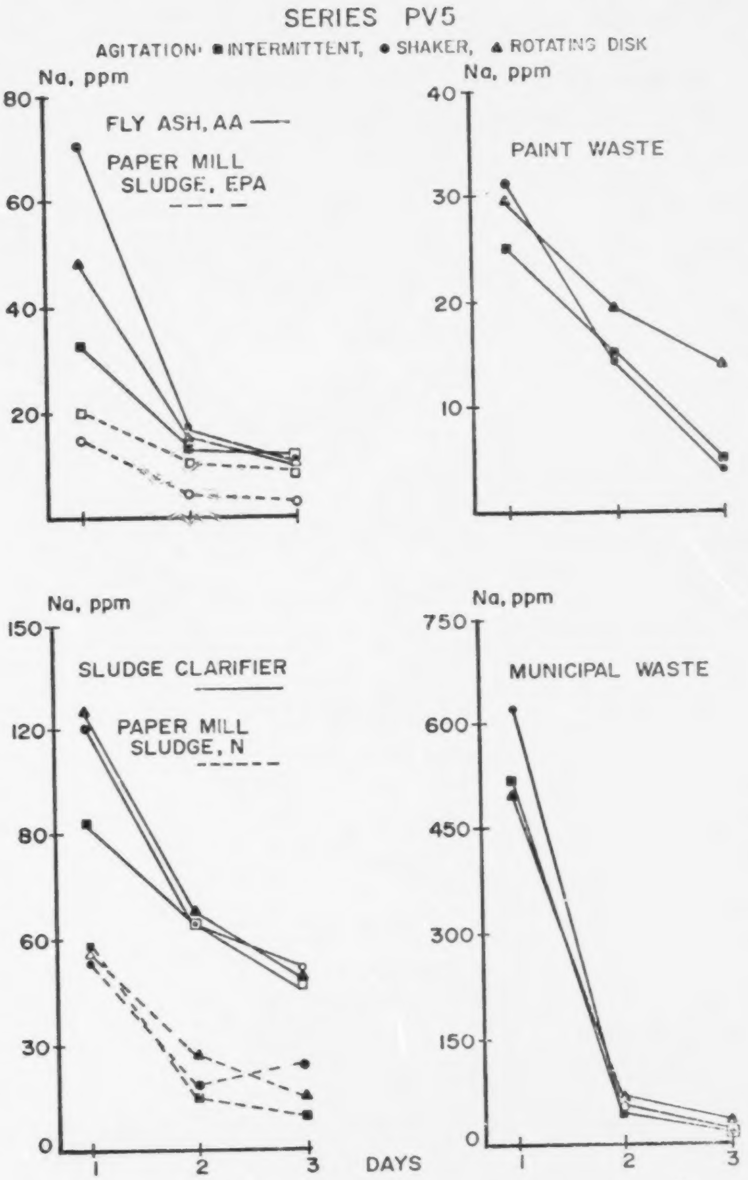

Figure A-45. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Na.

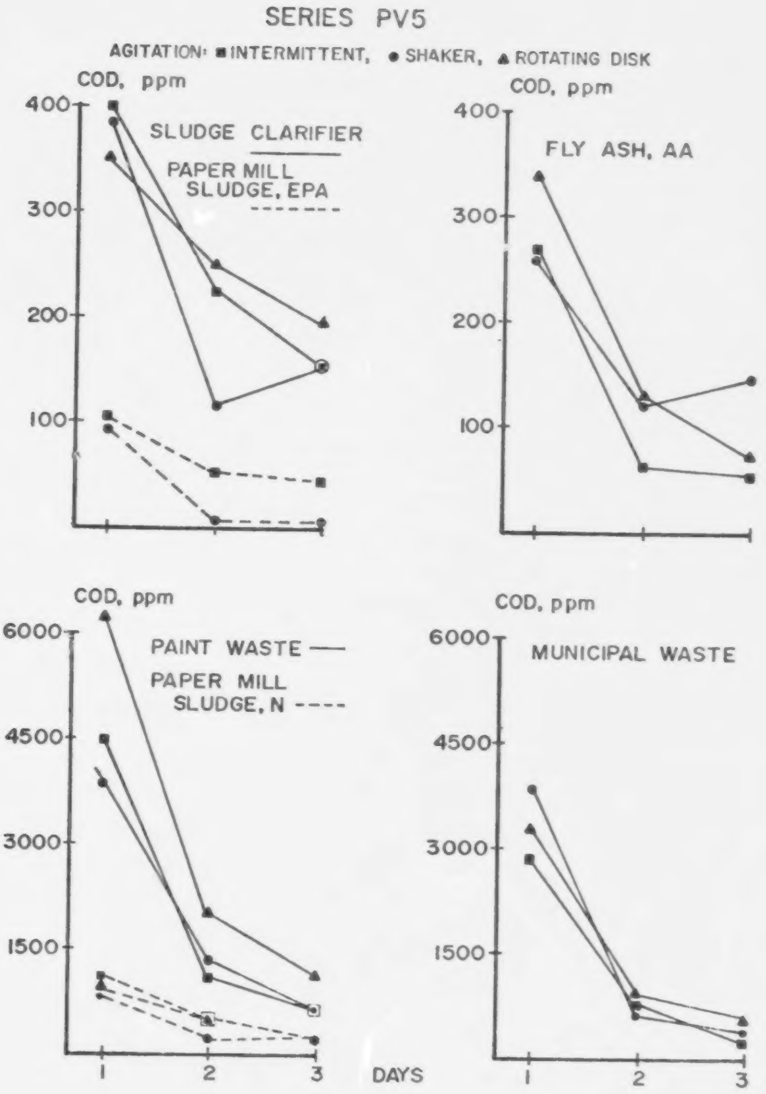

Figure A-46. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. COD.

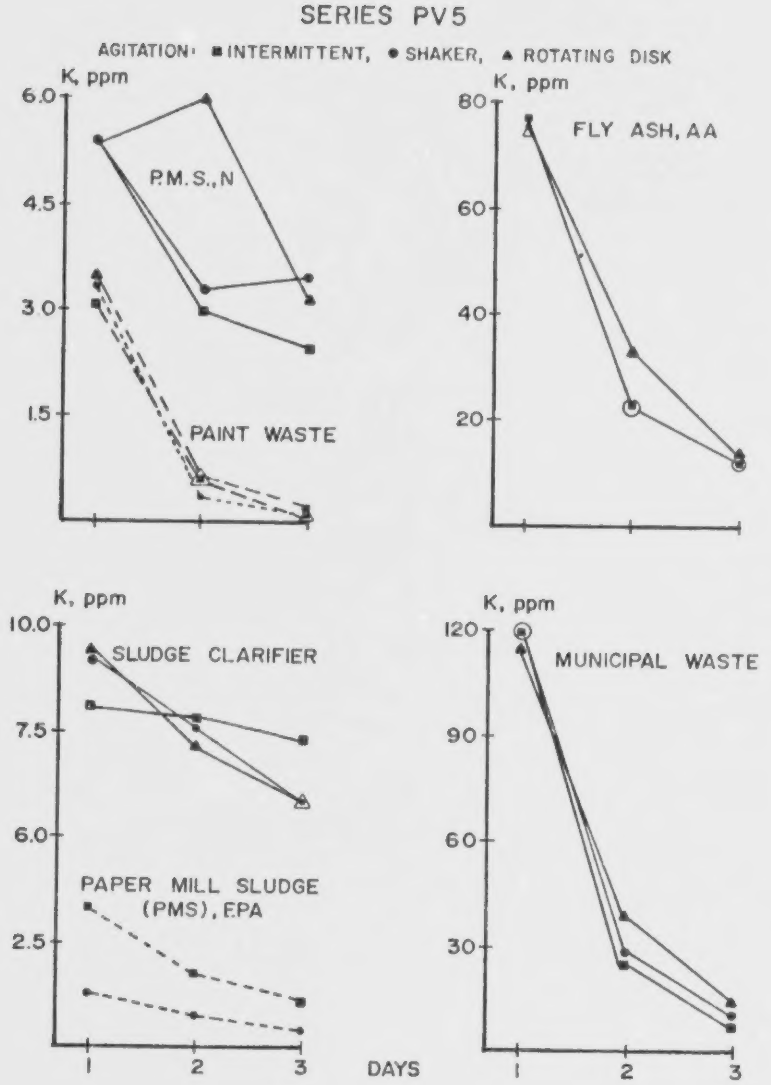

Figure A-47. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. K.

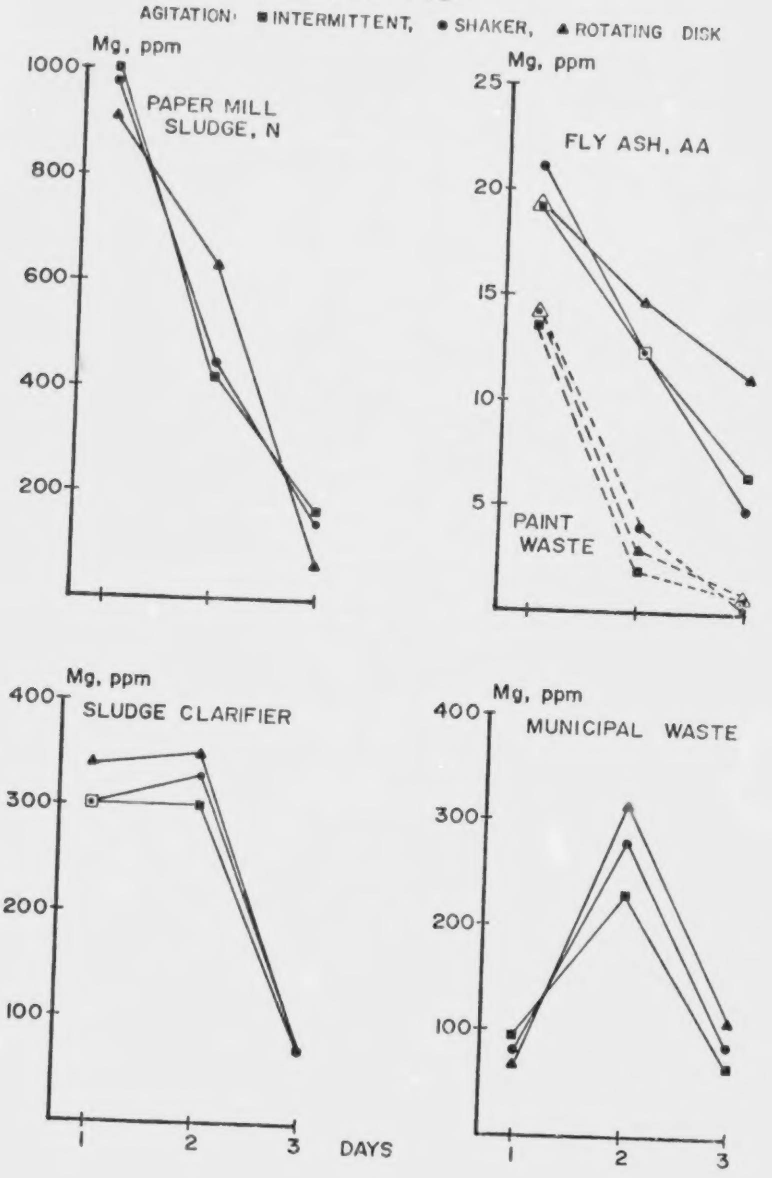

Figure A-48. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Mg.

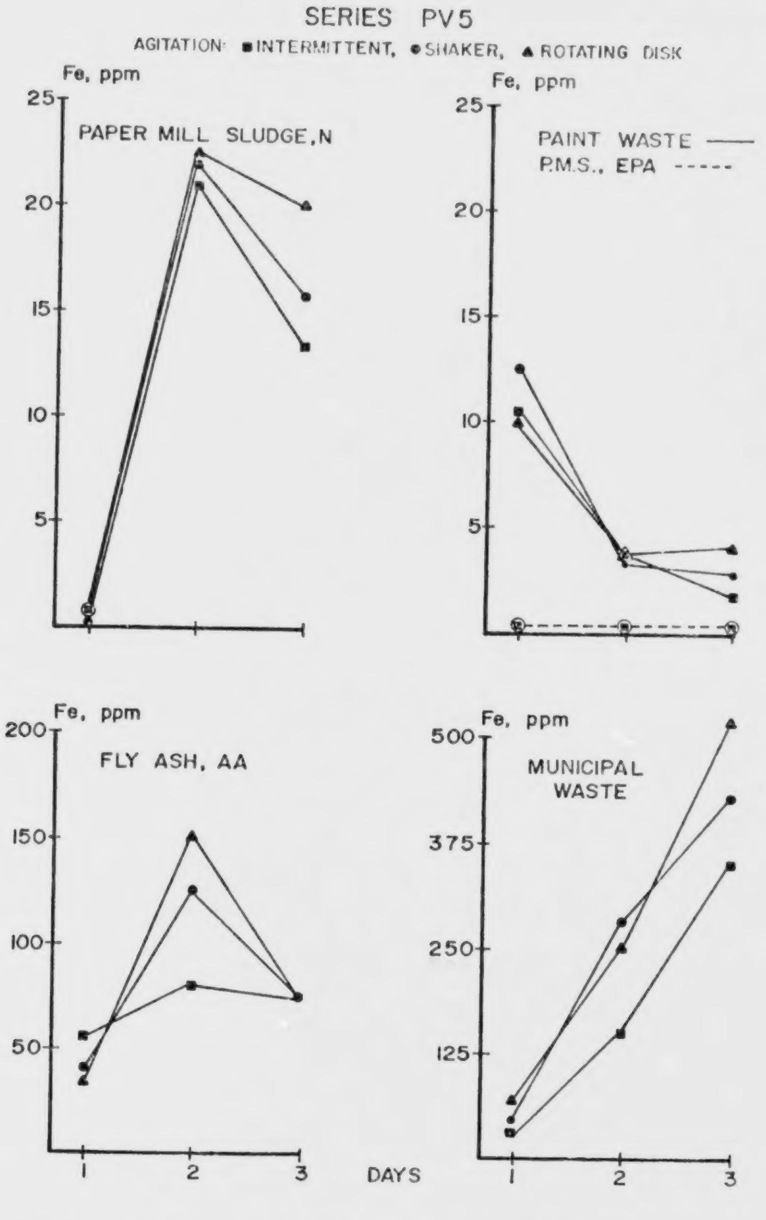

Figure A-49. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Fe.

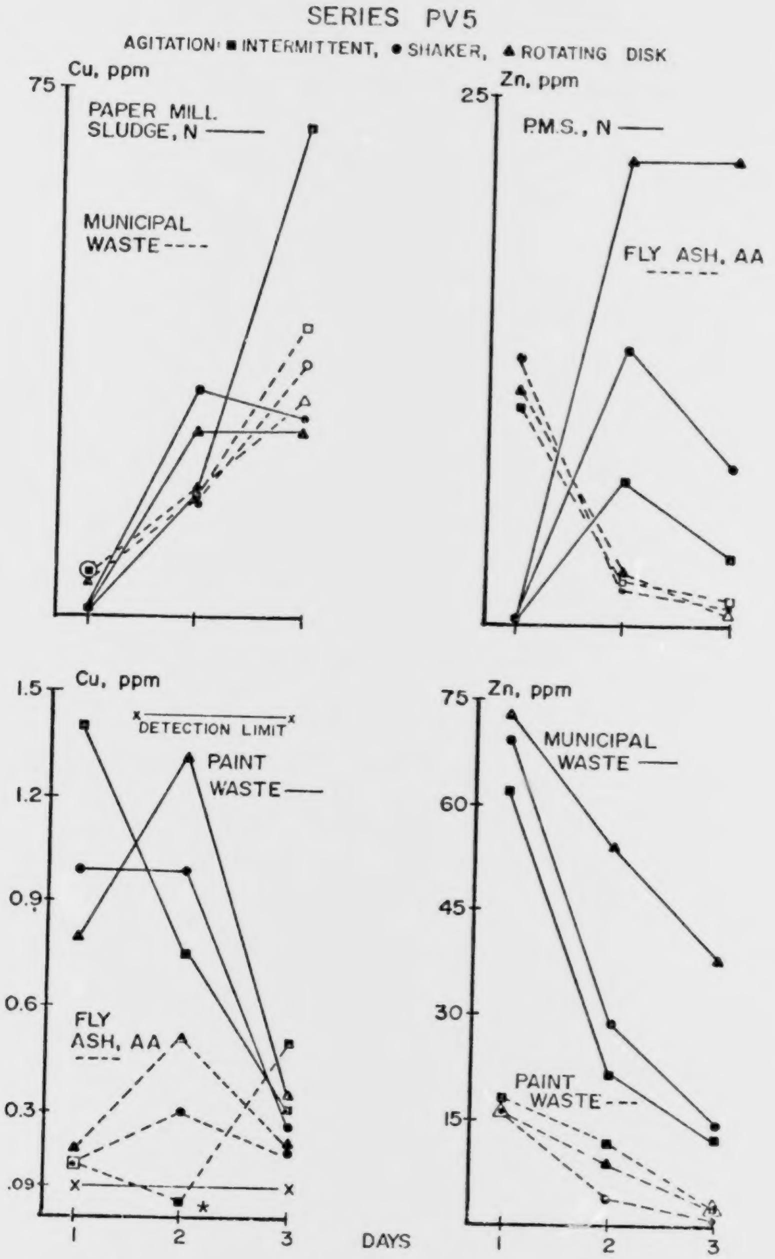

Figure A-50. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Cu and Zn.

*Approximate value.

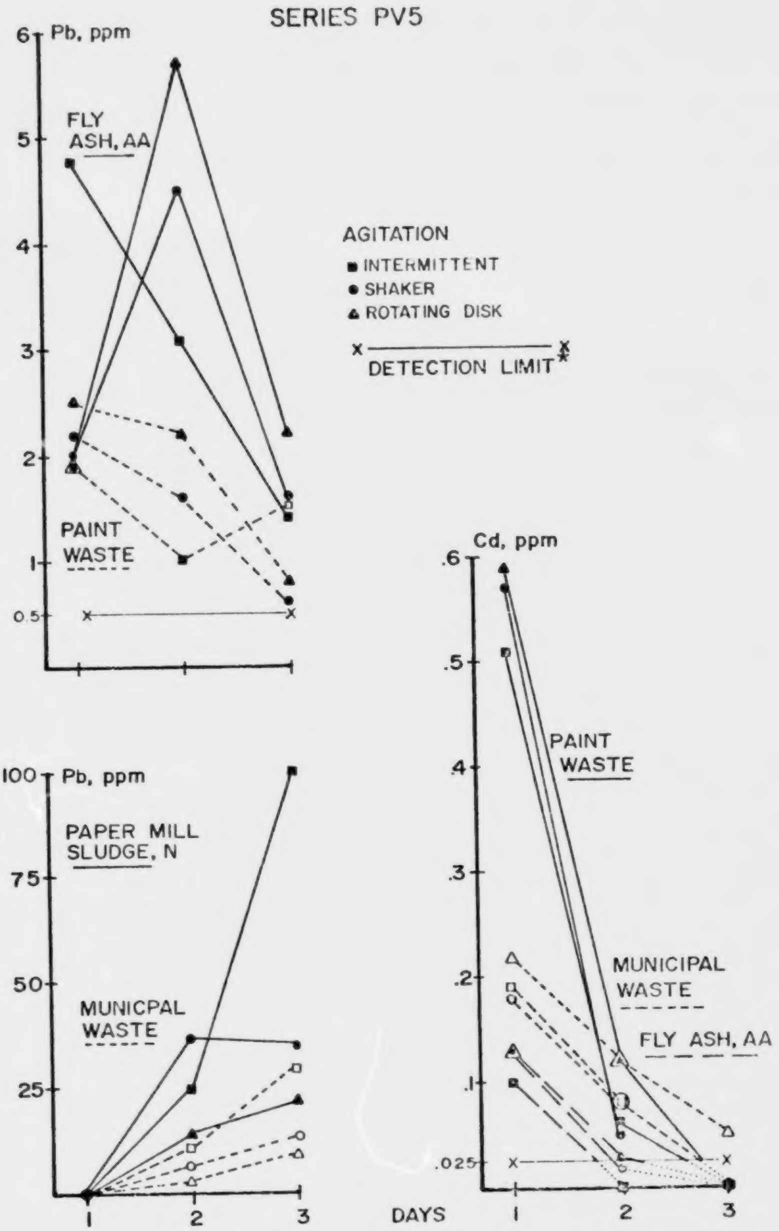

Figure A-51. Test PV5 comparing different agitation techniques on several wastes with 0.1N H_2SO_4. Pb and Cd.

*All points below detection limit approximate.

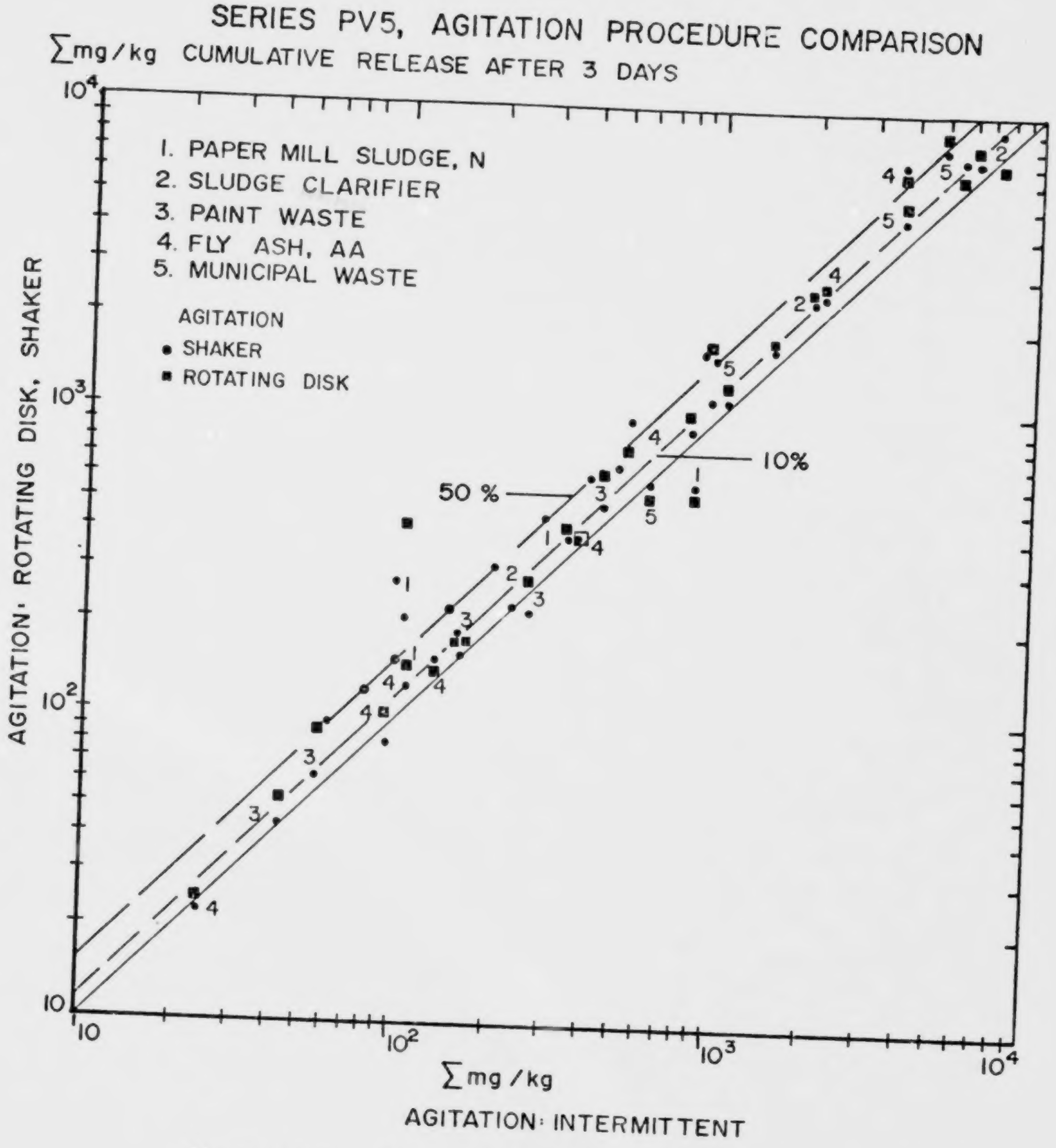

Figure A-52. Test PV5 comparing the cumulative release of all measured parameters after 3 elutions using the rotating disc and intermittent shaking agitation techniques.

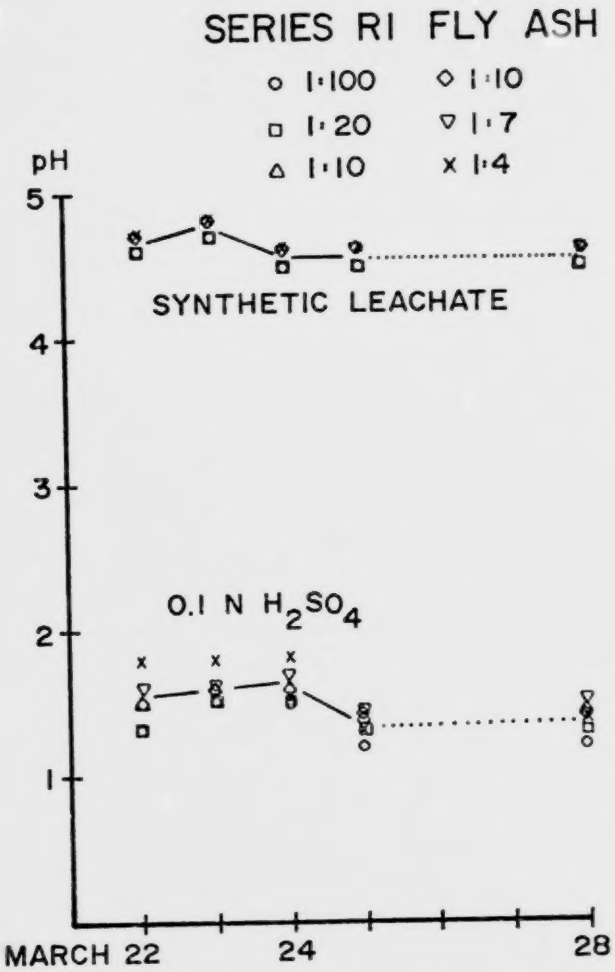

Figure A-53. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and synthetic leachate and 0.1N H_2SO_4. pH.

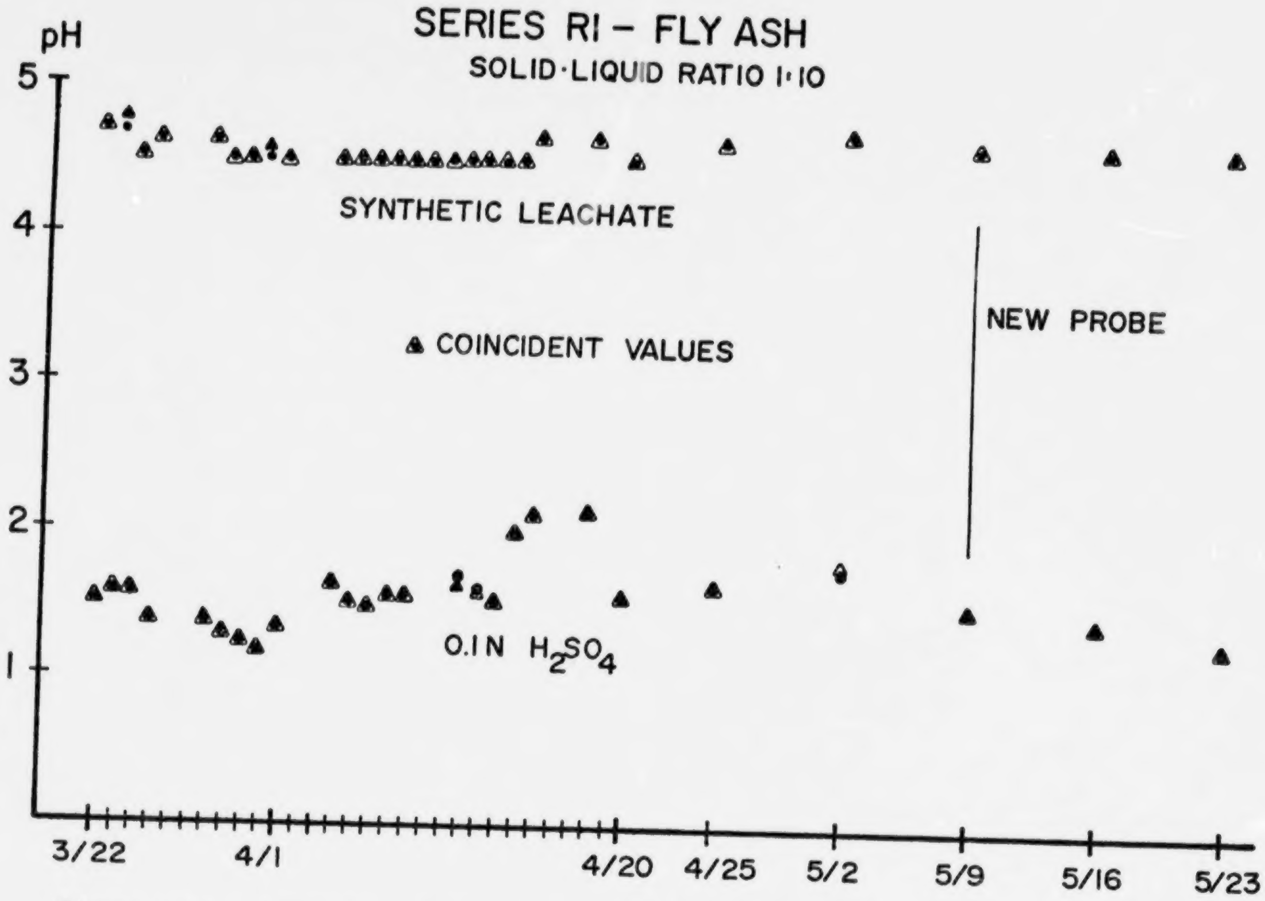

Figure A-54. Expansion of Figure A-53 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

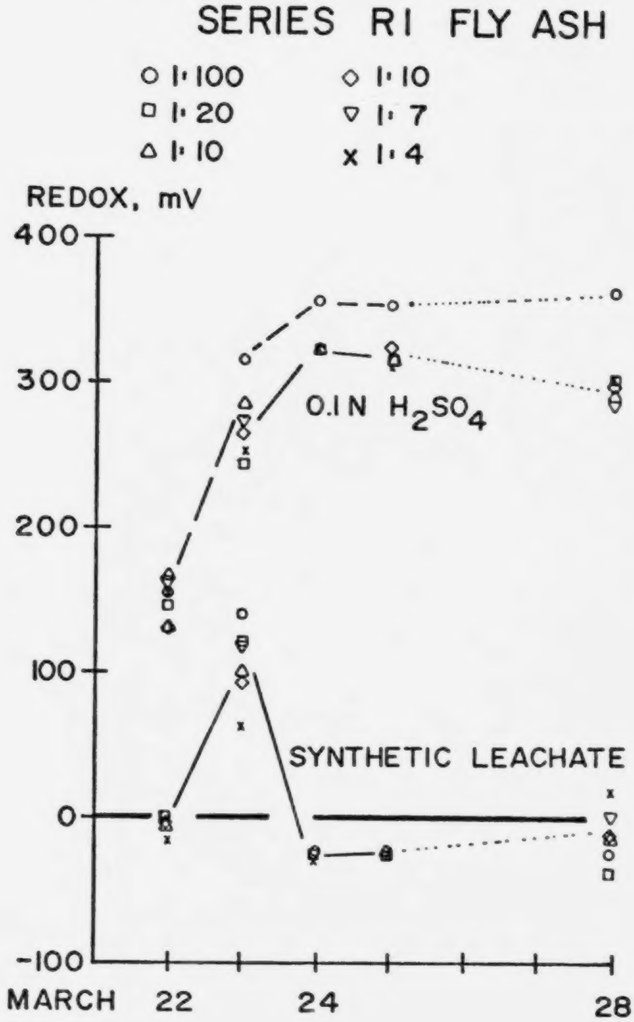

Figure A-55. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and synthetic leachate and 0.1N H_2SO_4. Redox.

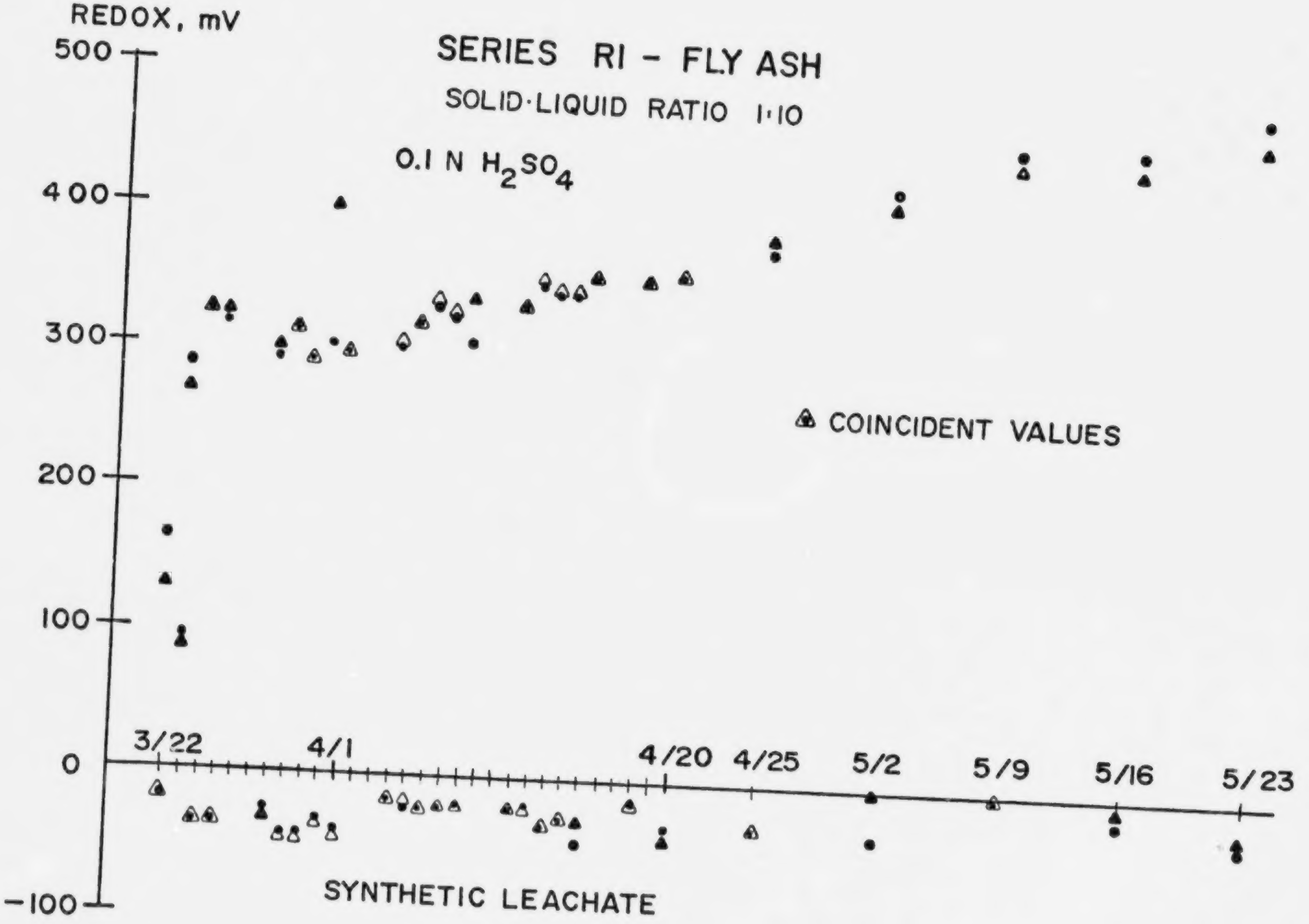

Figure A-56. Expansion of Figure A-55 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

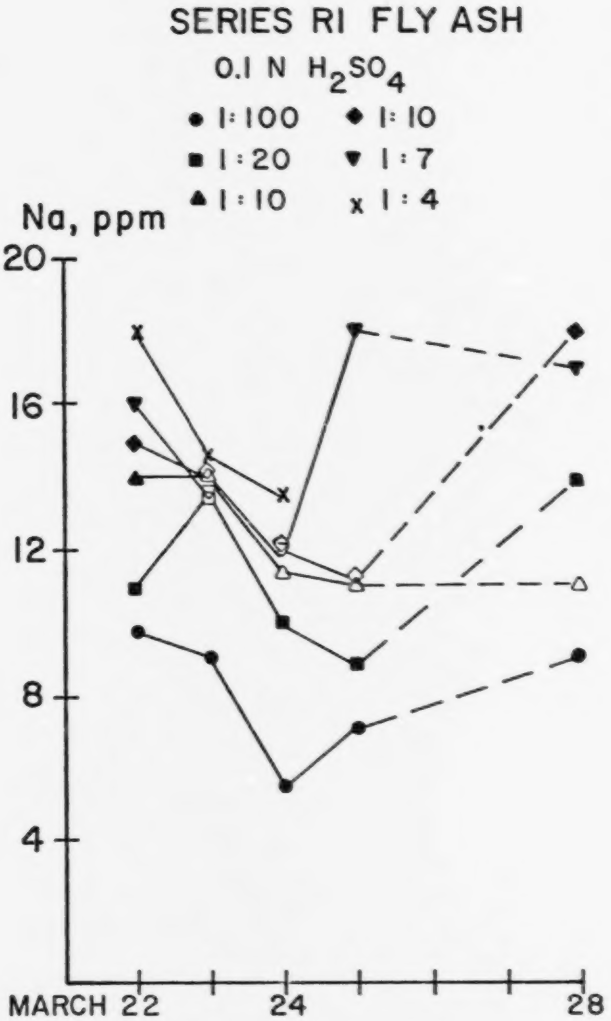

Figure A-57. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and 0.1N H_2SO_4. Na.

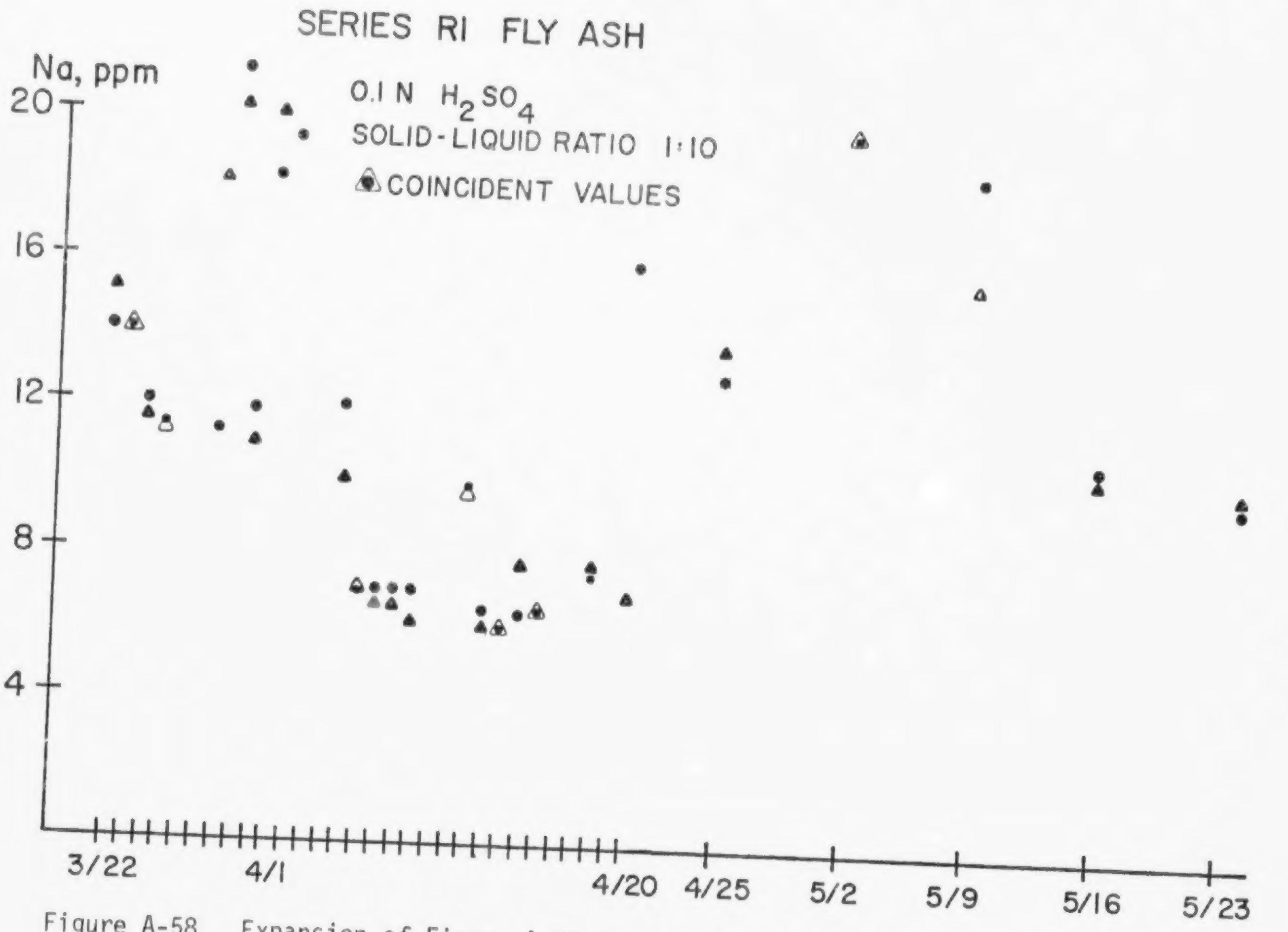

Figure A-58. Expansion of Figure A-57 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

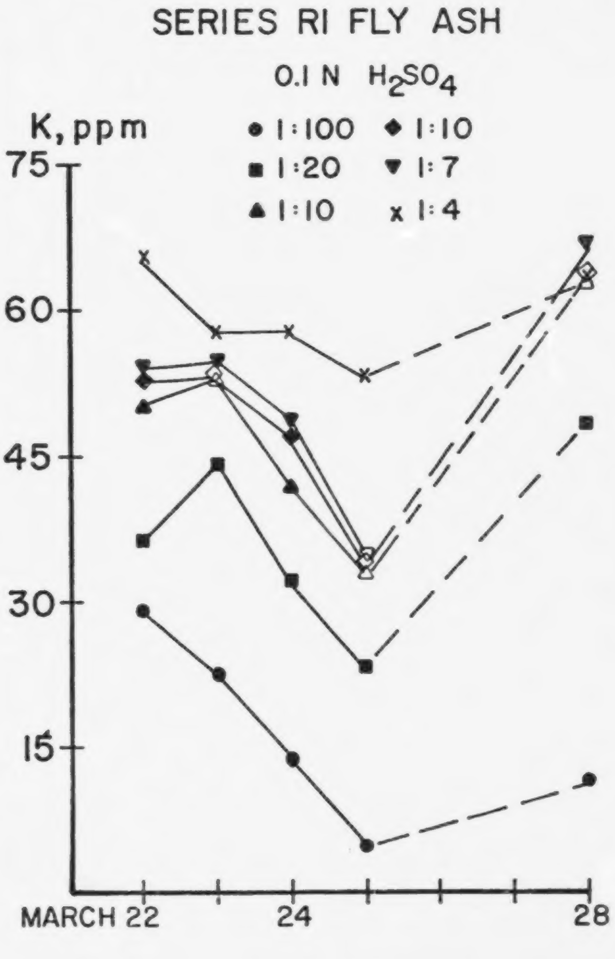

Figure A-59. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and 0.1N H_2SO_4. K.

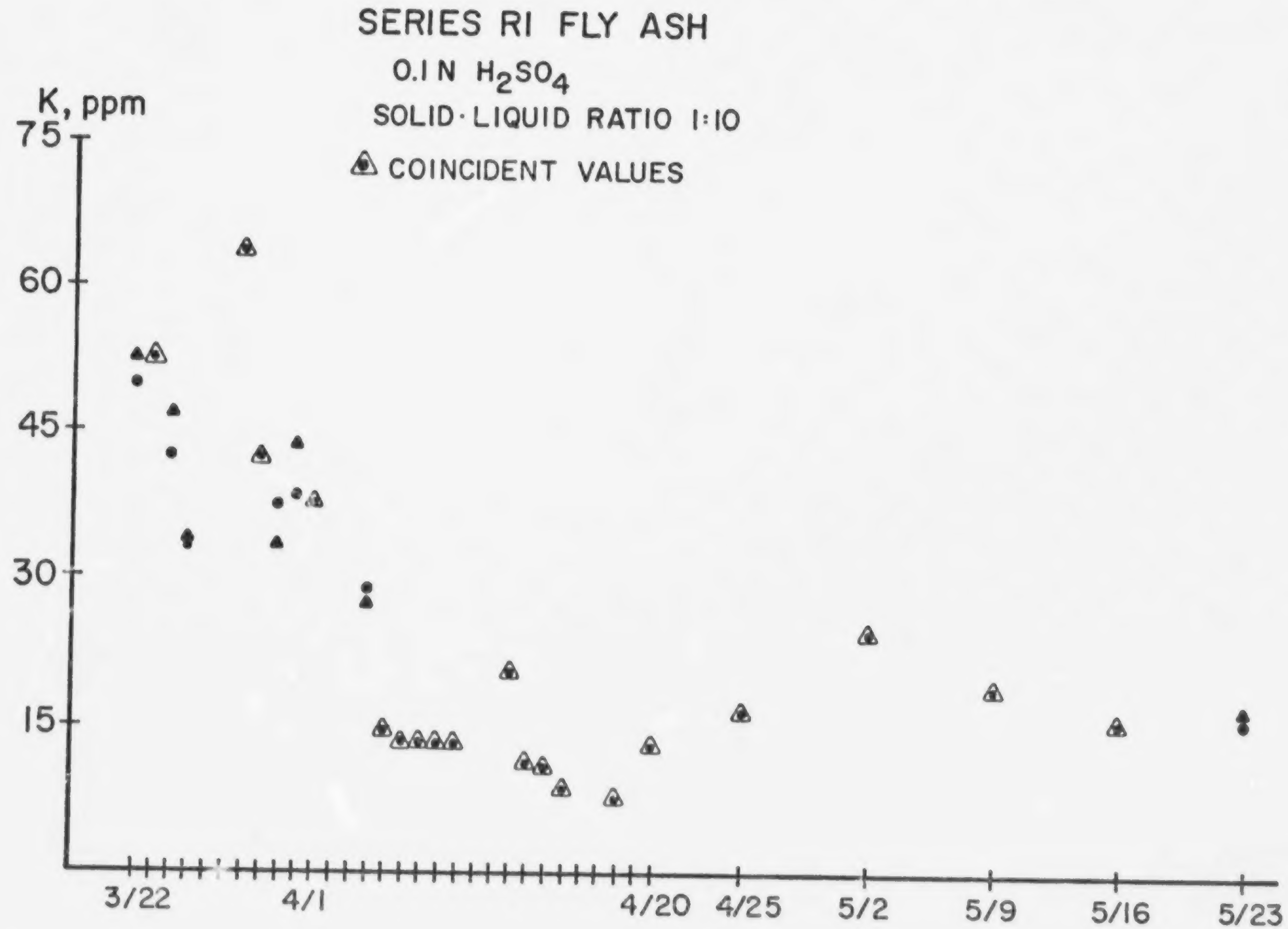

Figure A-60. Expansion of Figure A-59 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

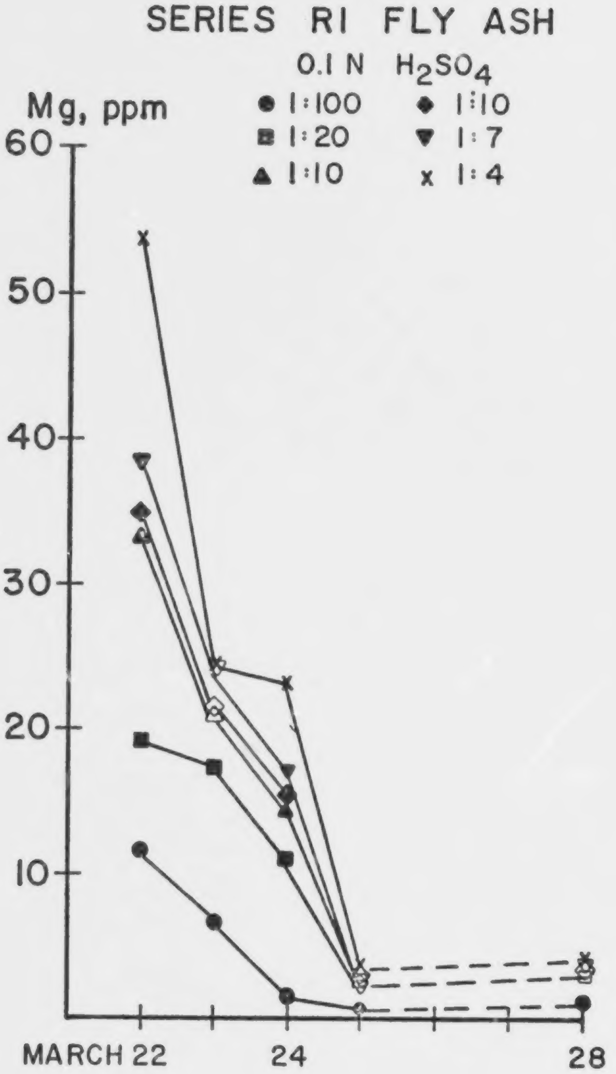

Figure A-61. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and 0.1N H_2SO_4. Mg.

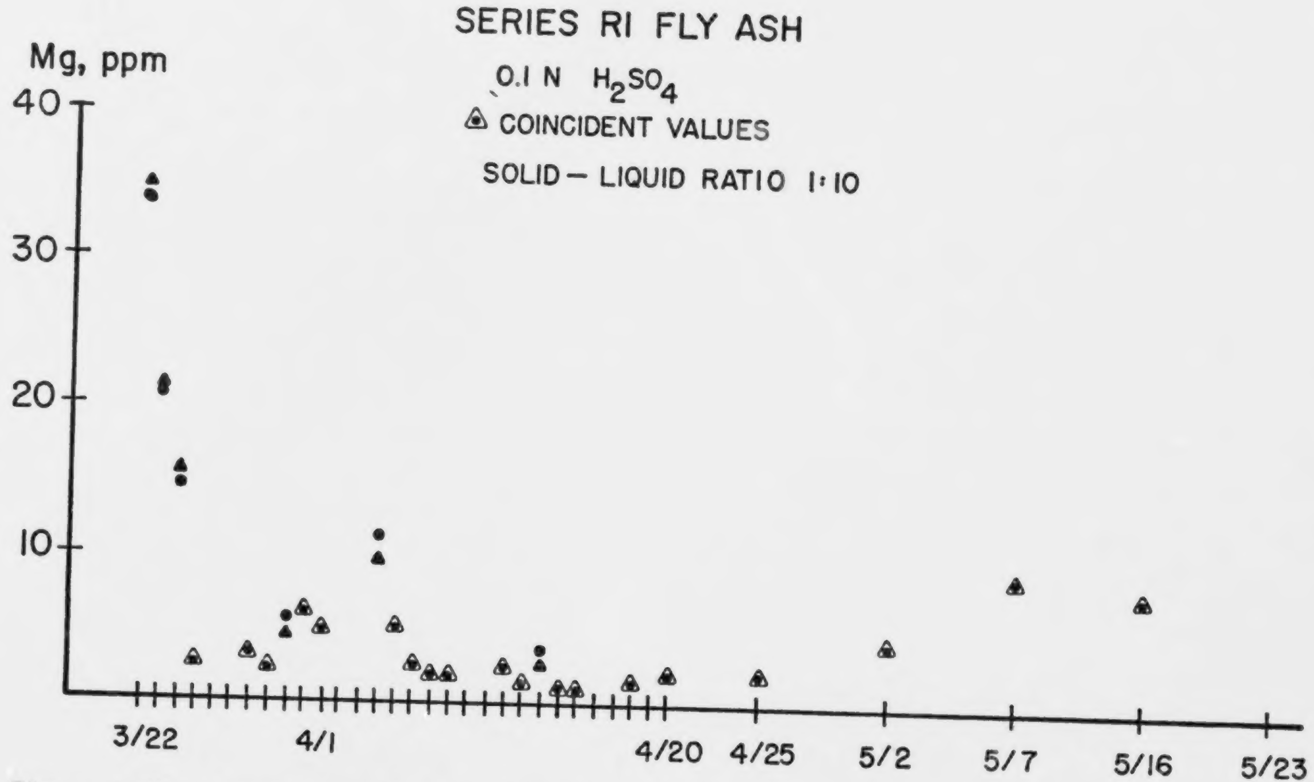

Figure A-62. Expansion of Figure A-61 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

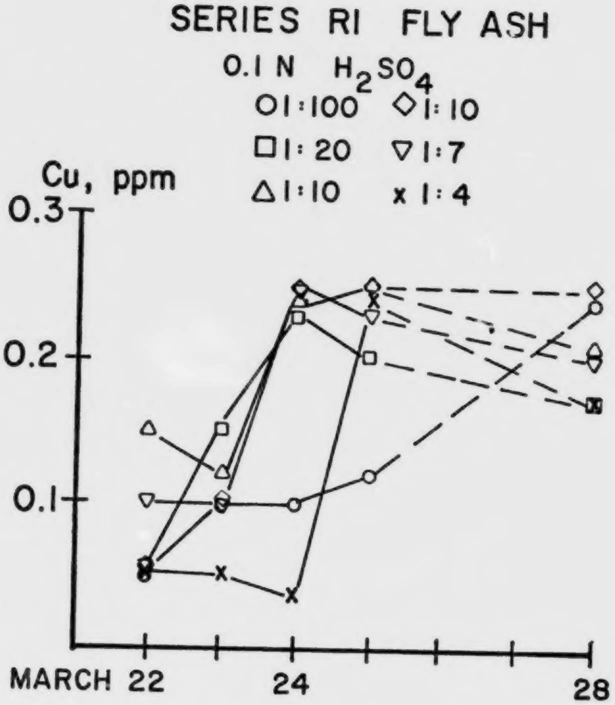

Figure A-63. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and 0.1N H_2SO_4. Cu.

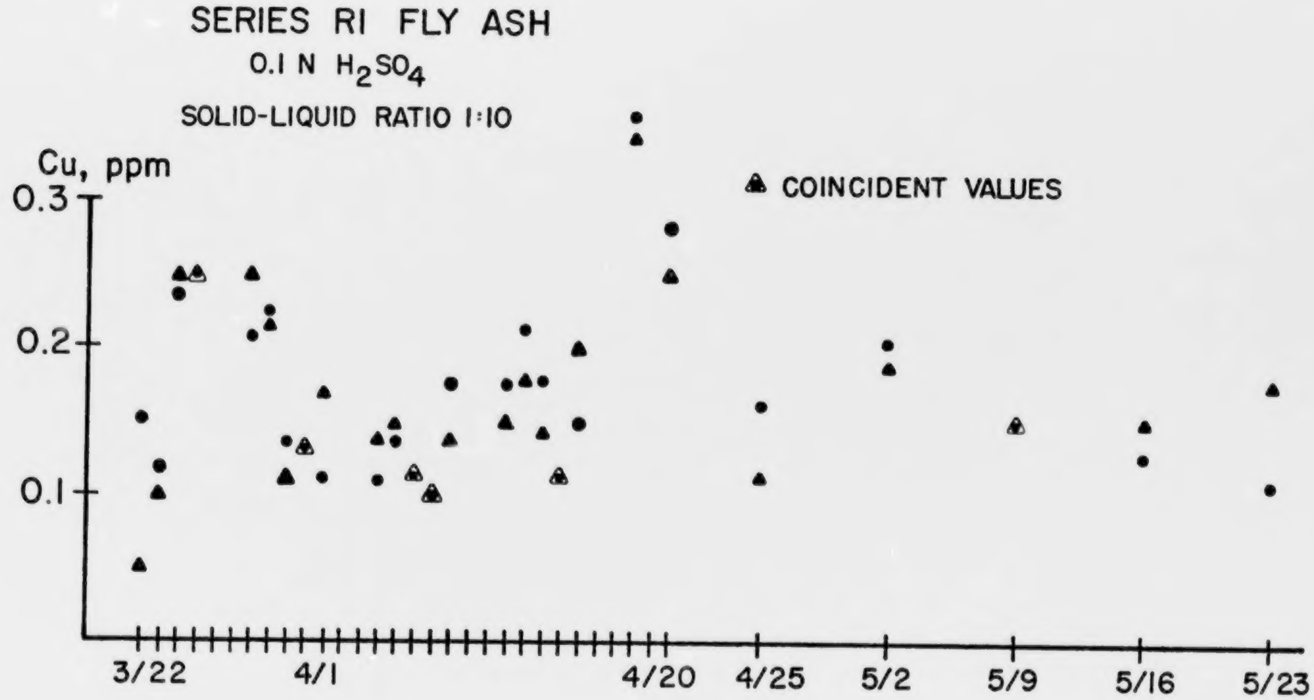

Figure A-64. Expansion of Figure A-63 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

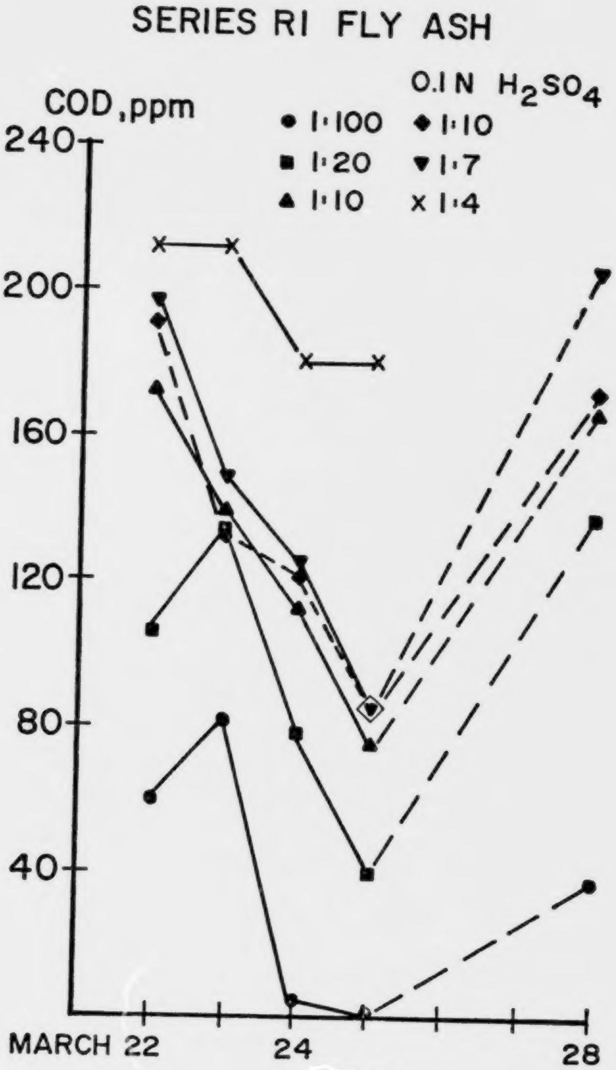

Figure A-65. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and 0.1N H_2SO_4. COD.

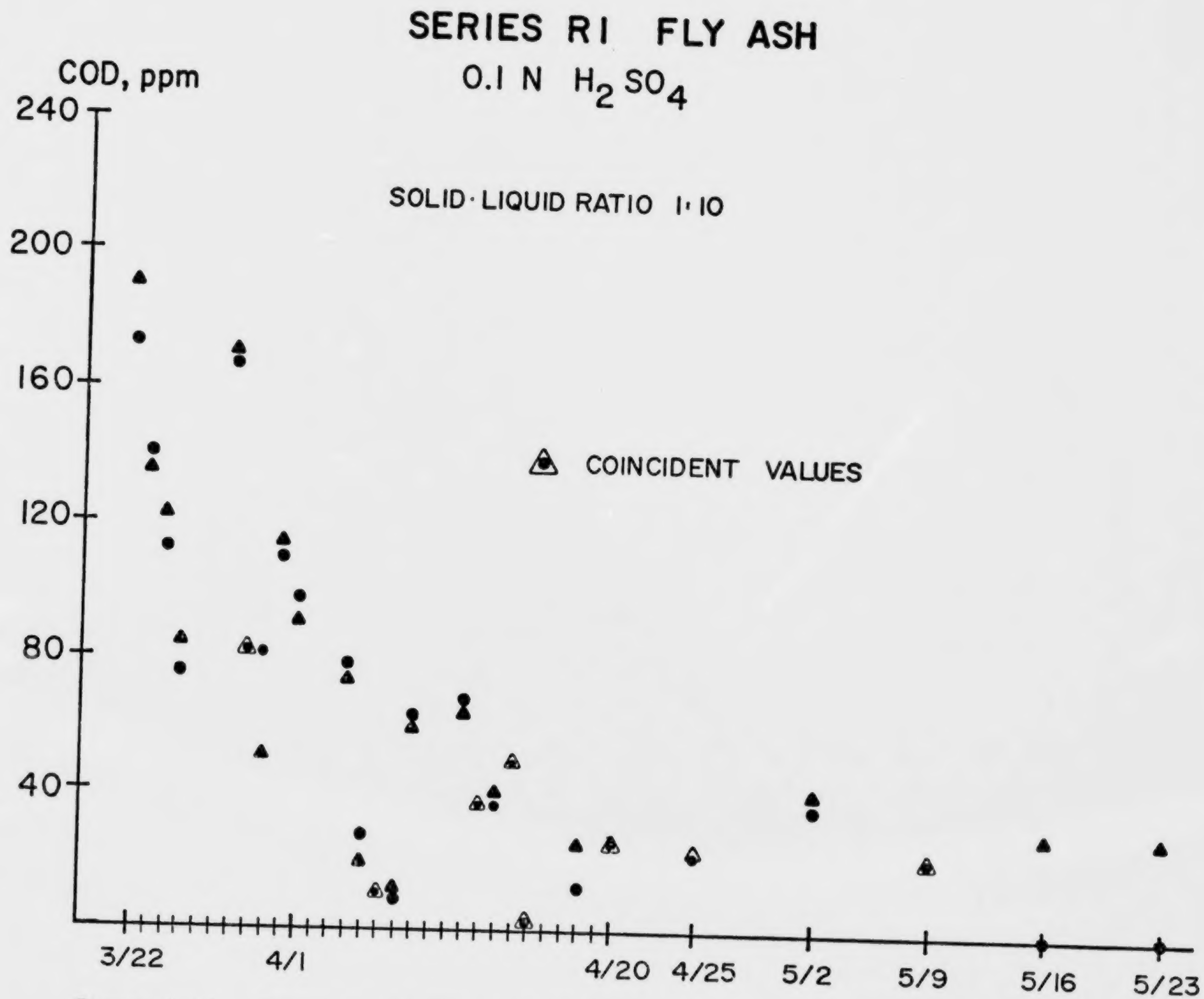

Figure A-66. Expansion of Figure A-65 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

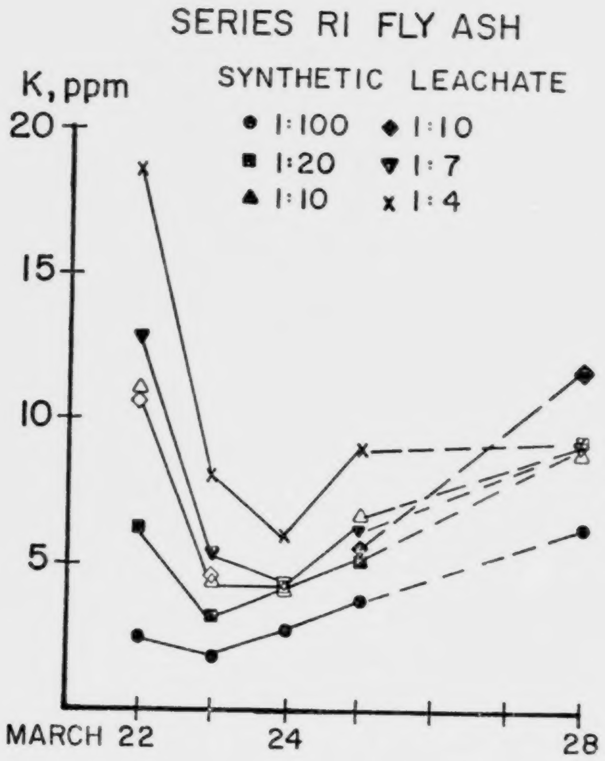

Figure A-67. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and synthetic leachate. K.

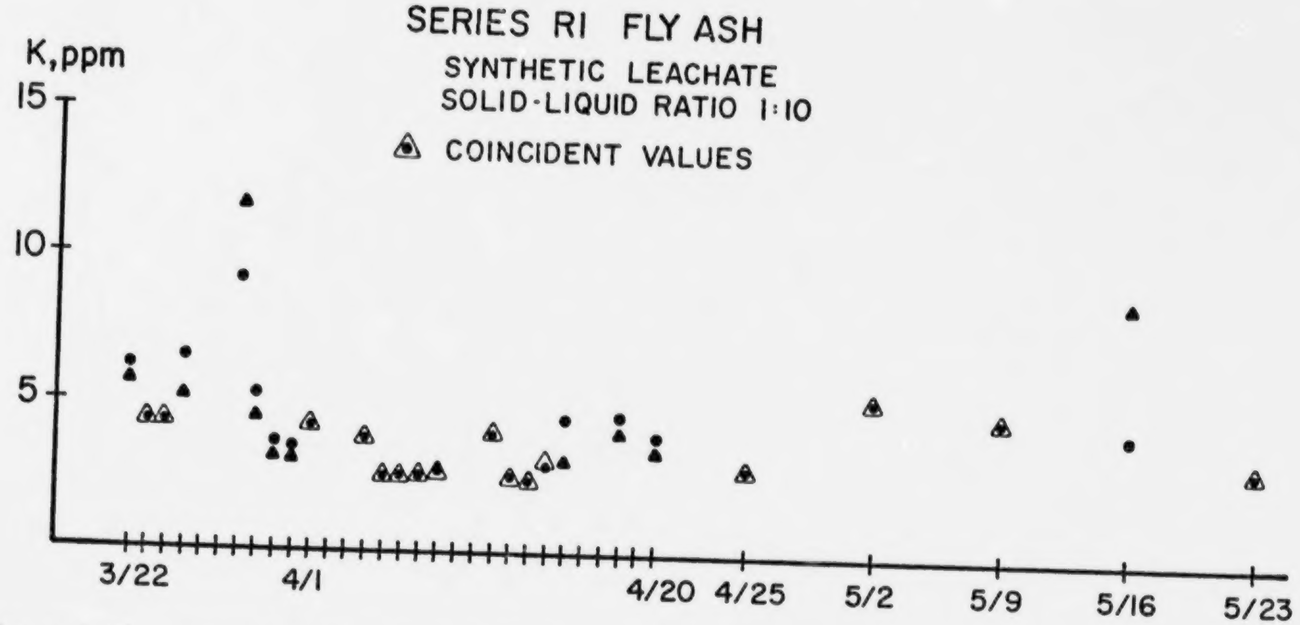

Figure A-68. Expansion of Figure A-67 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

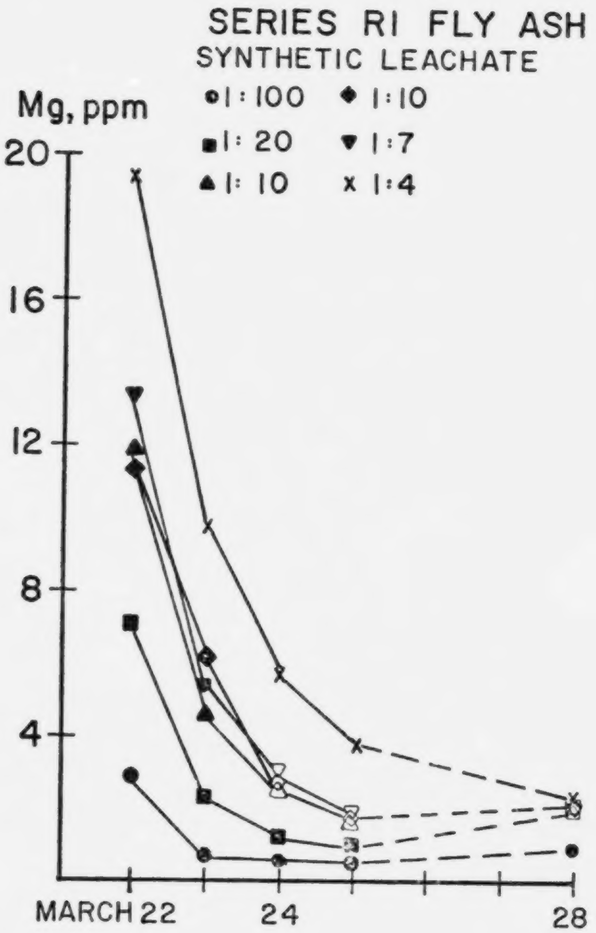

Figure A-69. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and synthetic leachate. Mg.

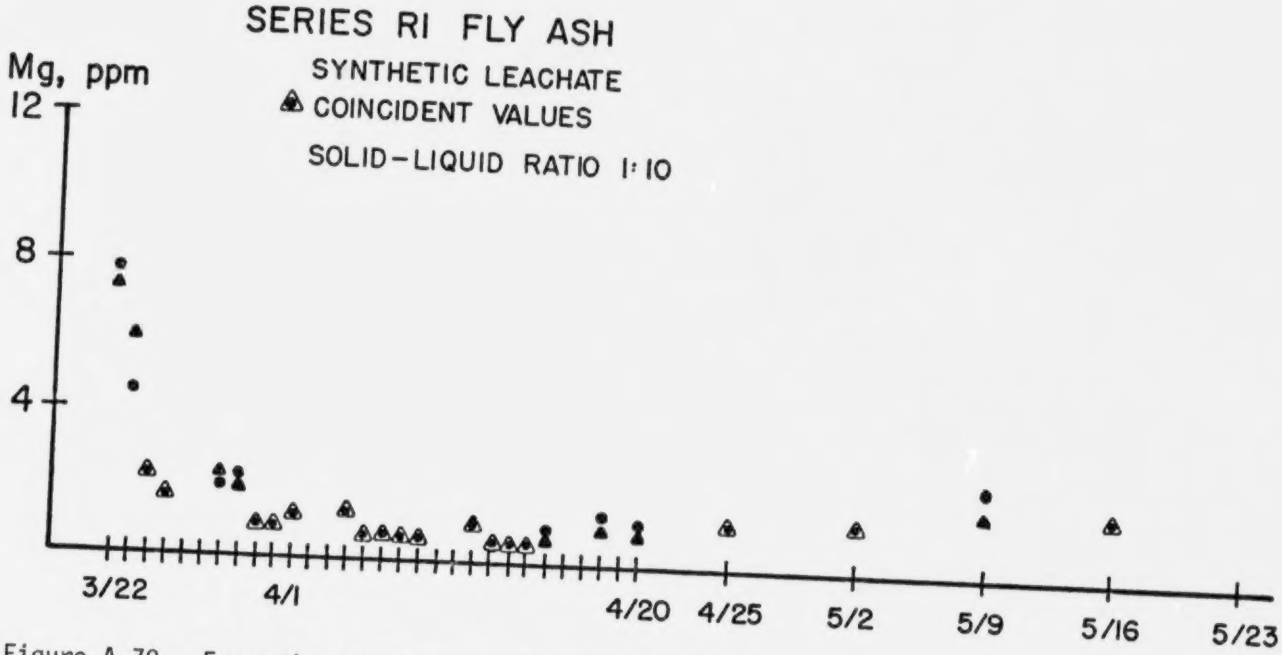

Figure A-70. Expansion of Figure A-69 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

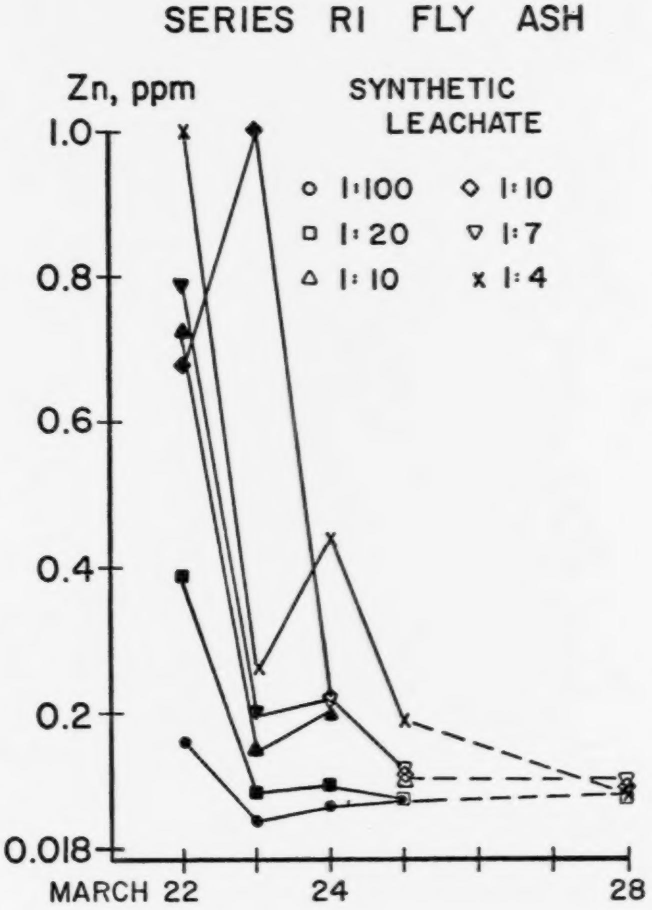

Figure A-71. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with fly ash and synthetic leachate. Zn.

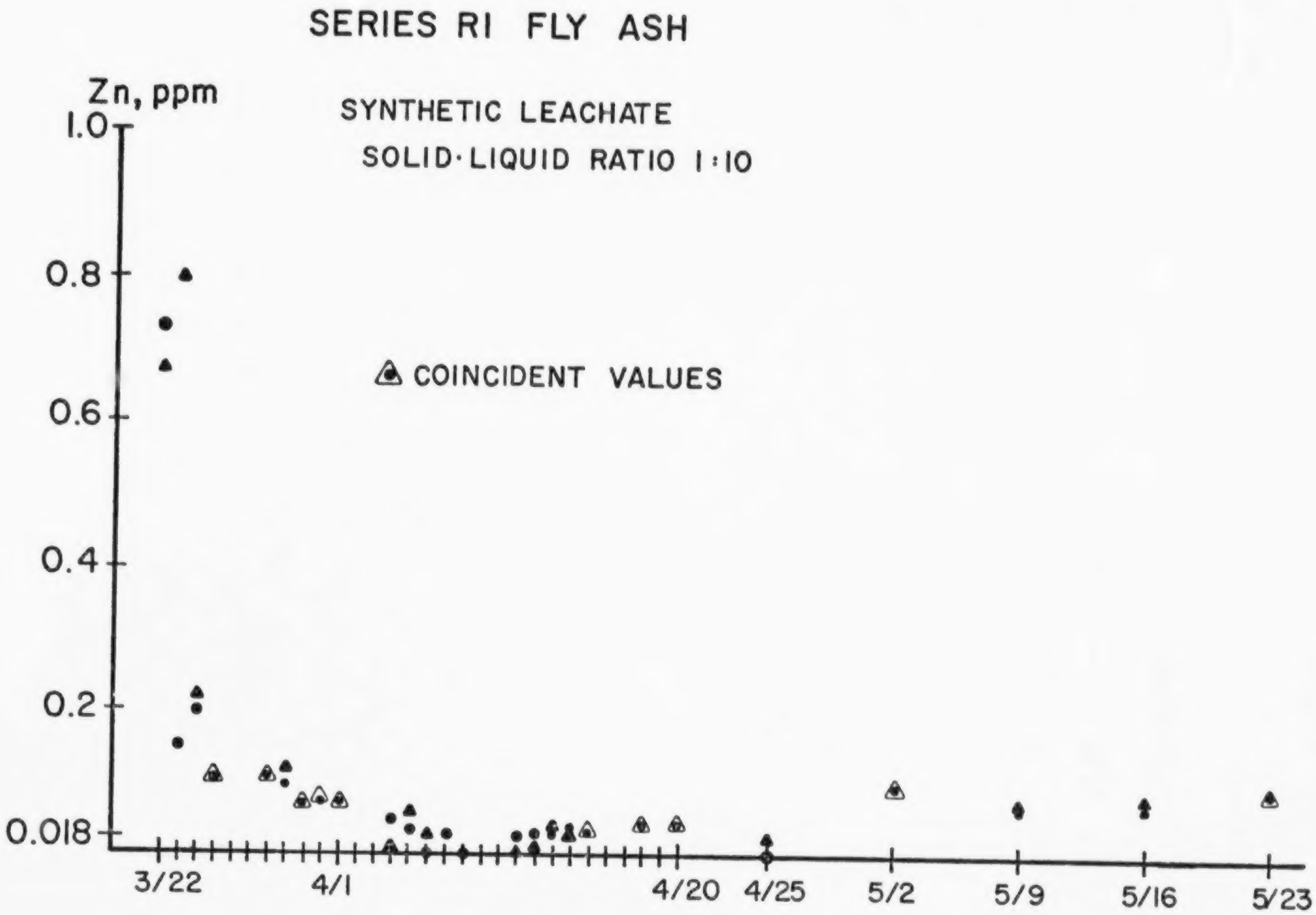

Figure A-72. Expansion of Figure A-71 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

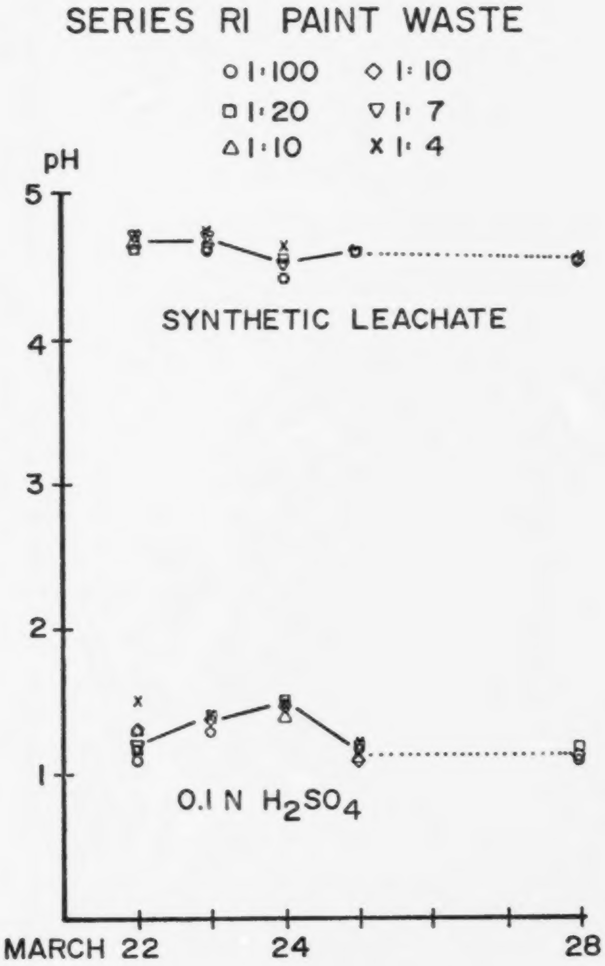

Figure A-73. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with paint waste and synthetic leachate and 0.1N H_2SO_4. pH.

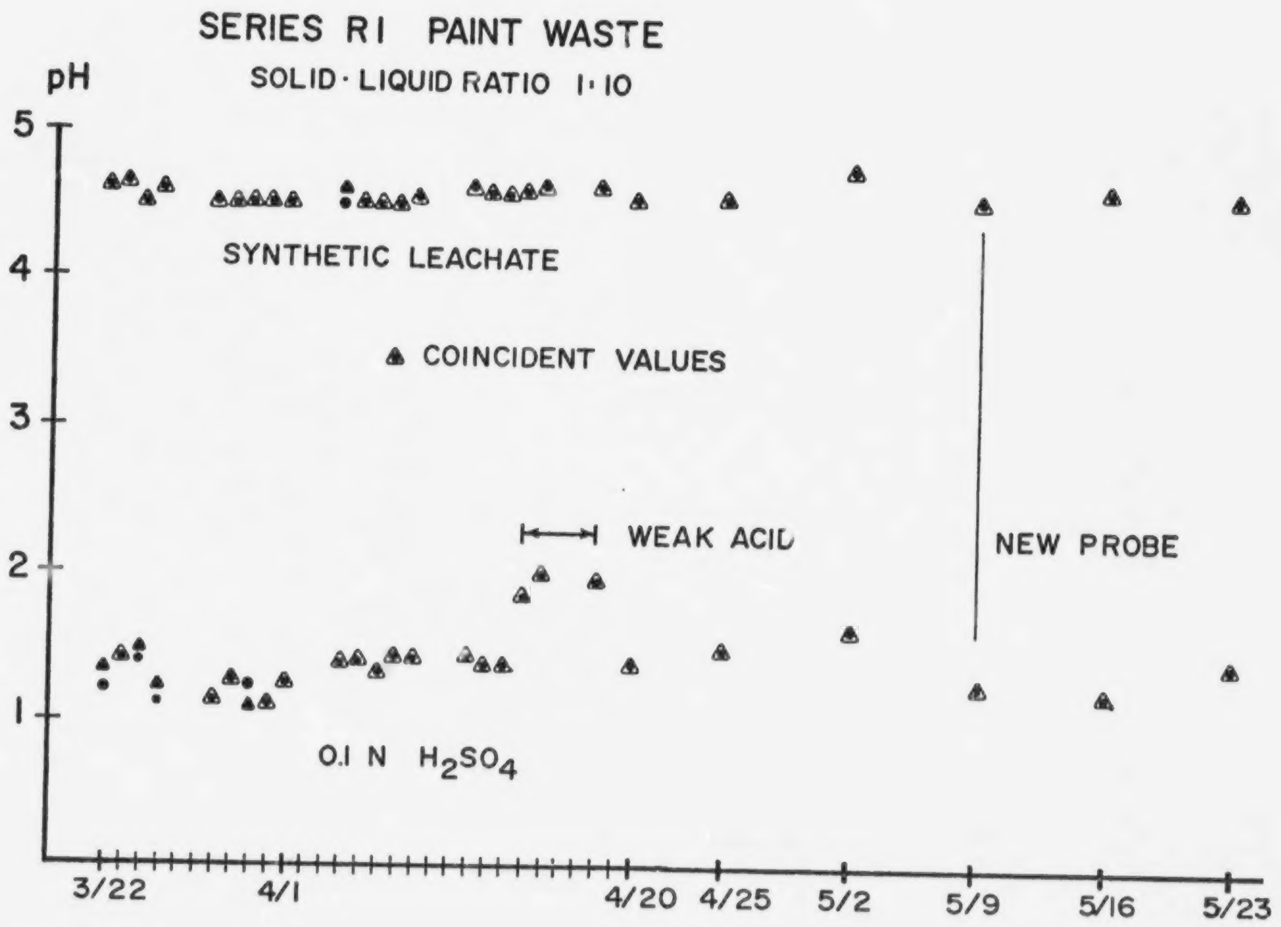

Figure A-74. Expansion of Figure A-73 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

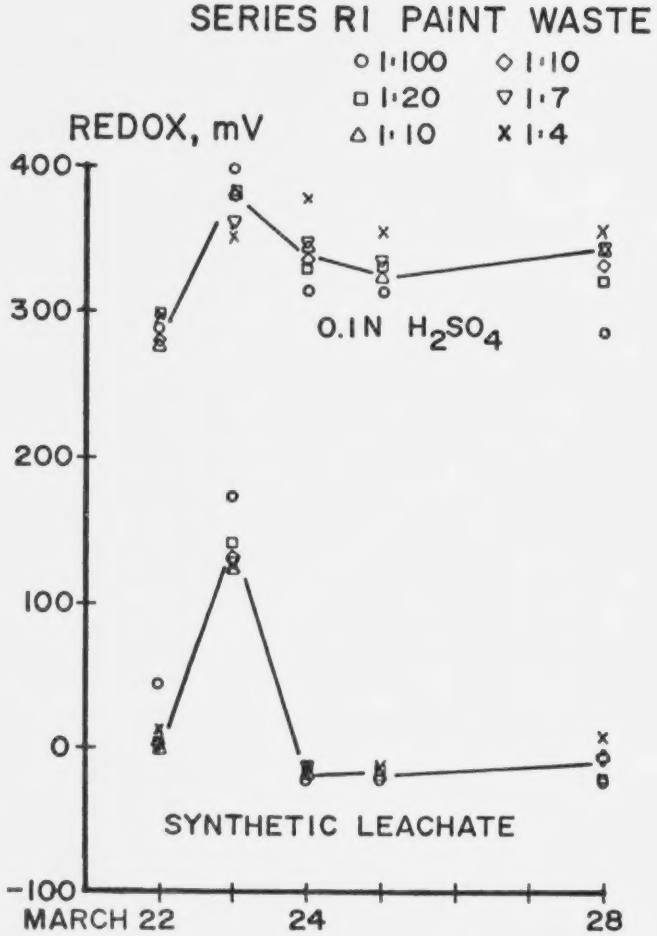

Figure A-75. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with paint waste and synthetic leachate and 0.1N H_2SO_4. Redox.

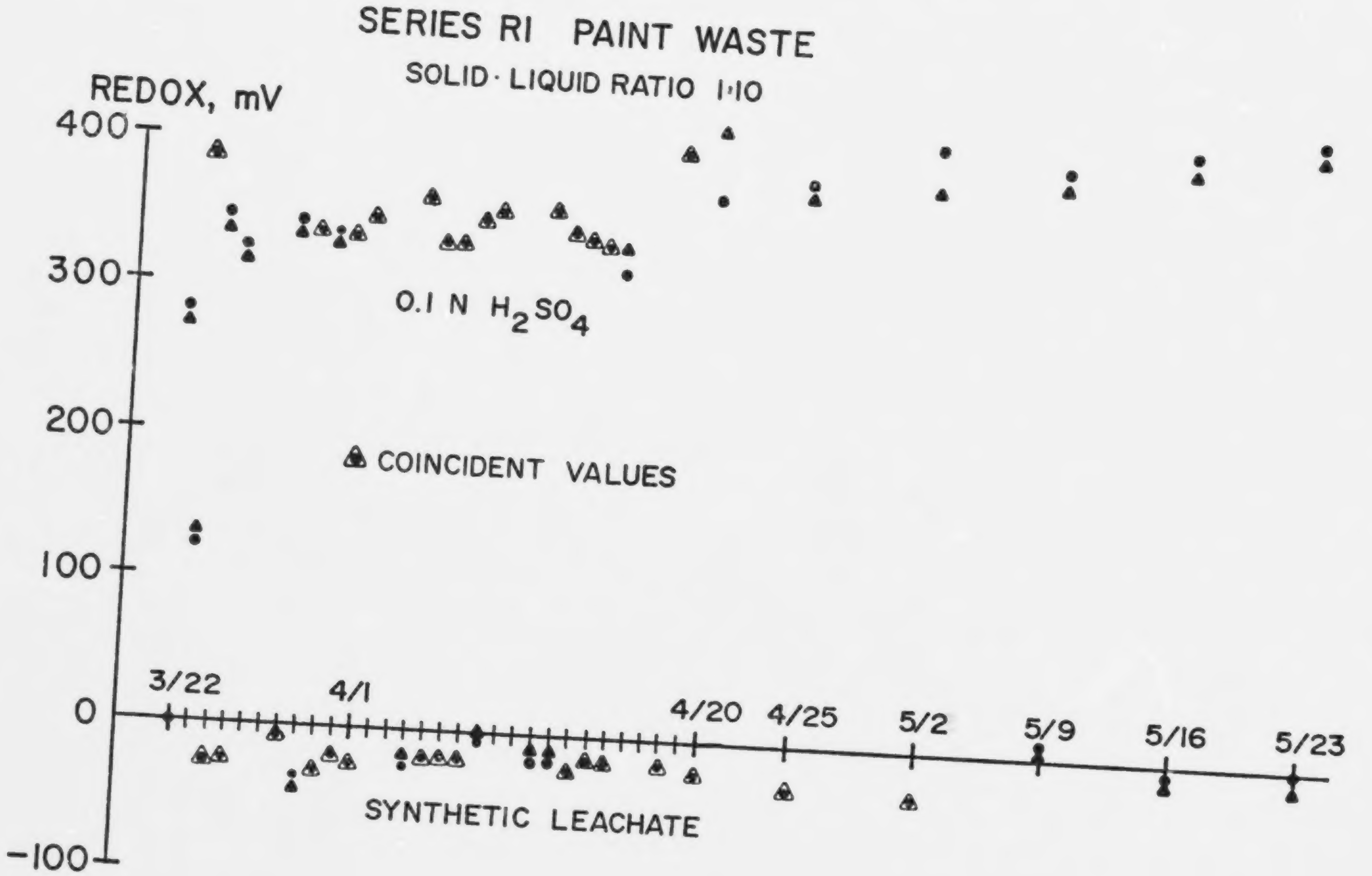

Figure A-76. Expansion of Figure A-75 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

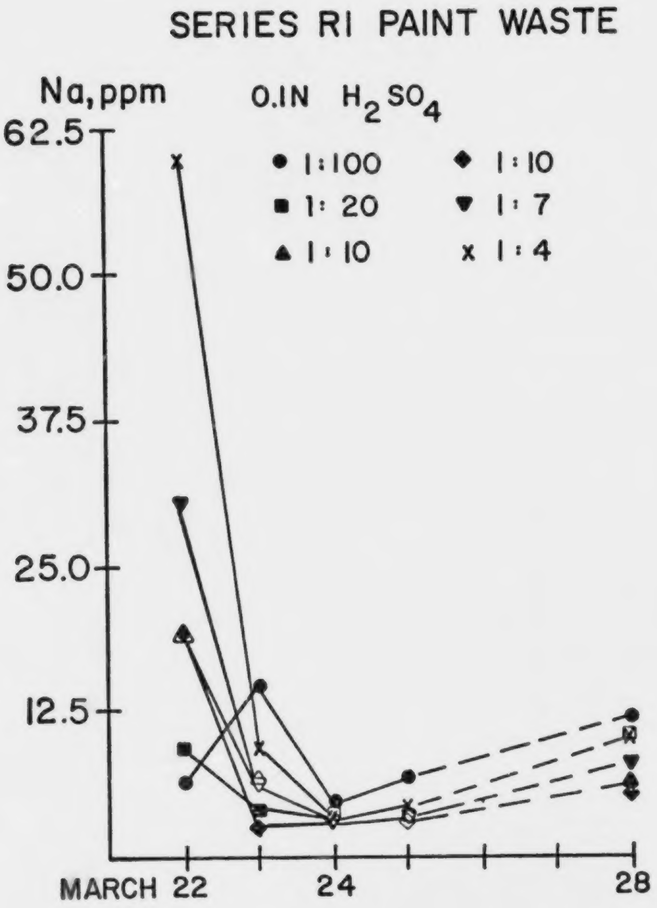

Figure A-77. Test R1 using procedure R to evaluate different solid-liquid ratios over five elutions with paint waste and 0.1N H_2SO_4. Na.

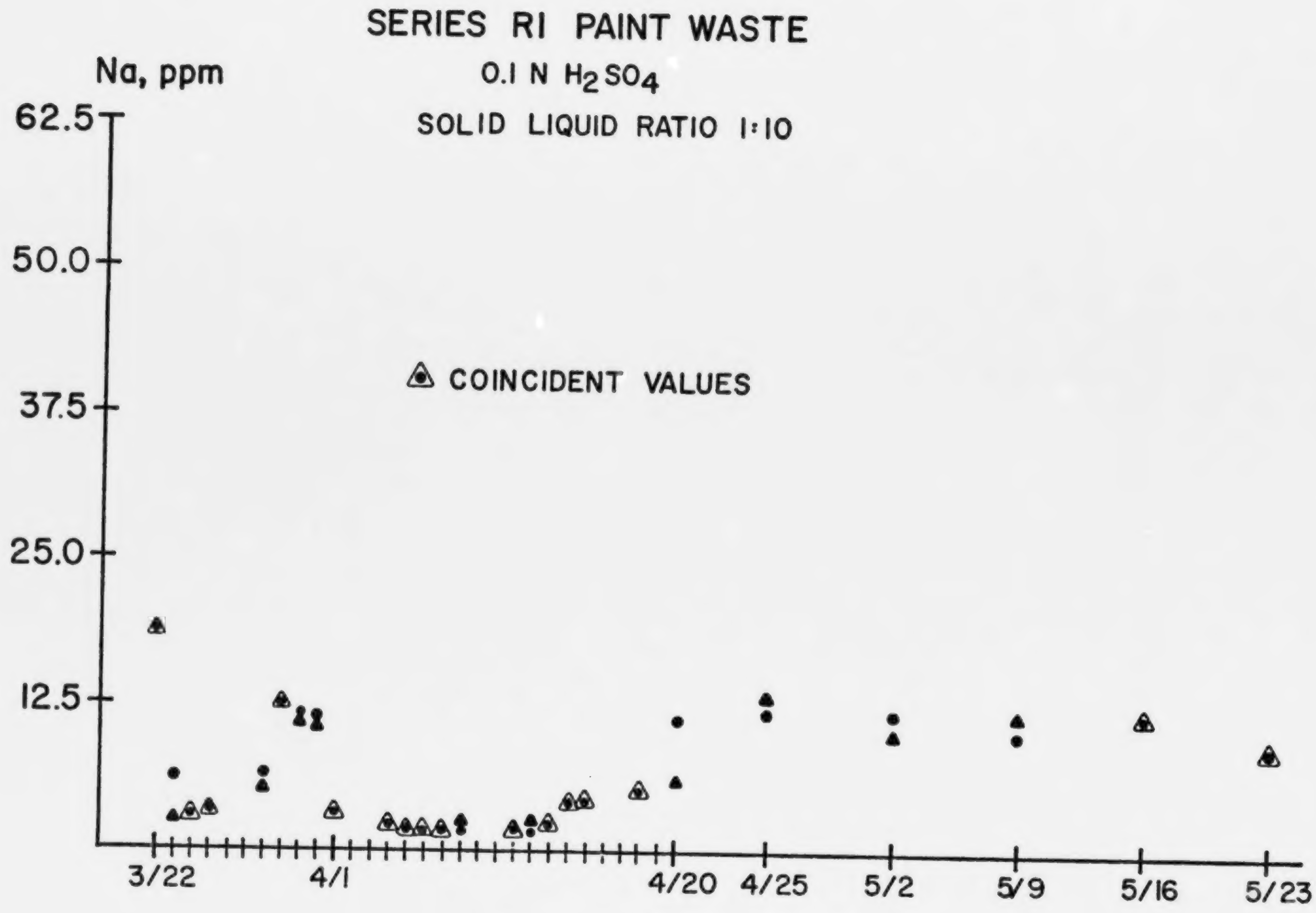

Figure A-78. Expansion of Figure A-77 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

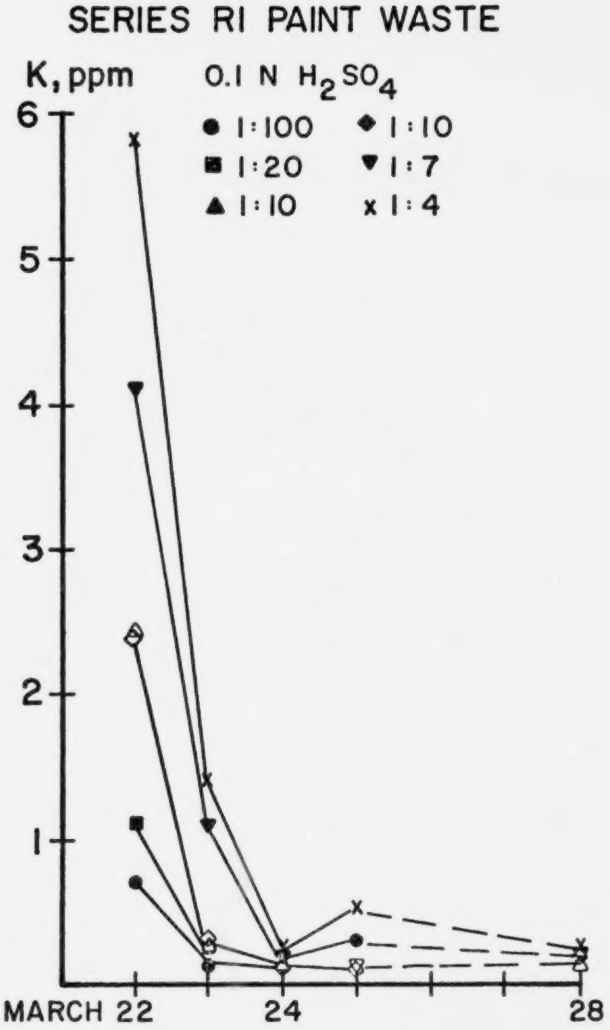

Figure A-79. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. K.

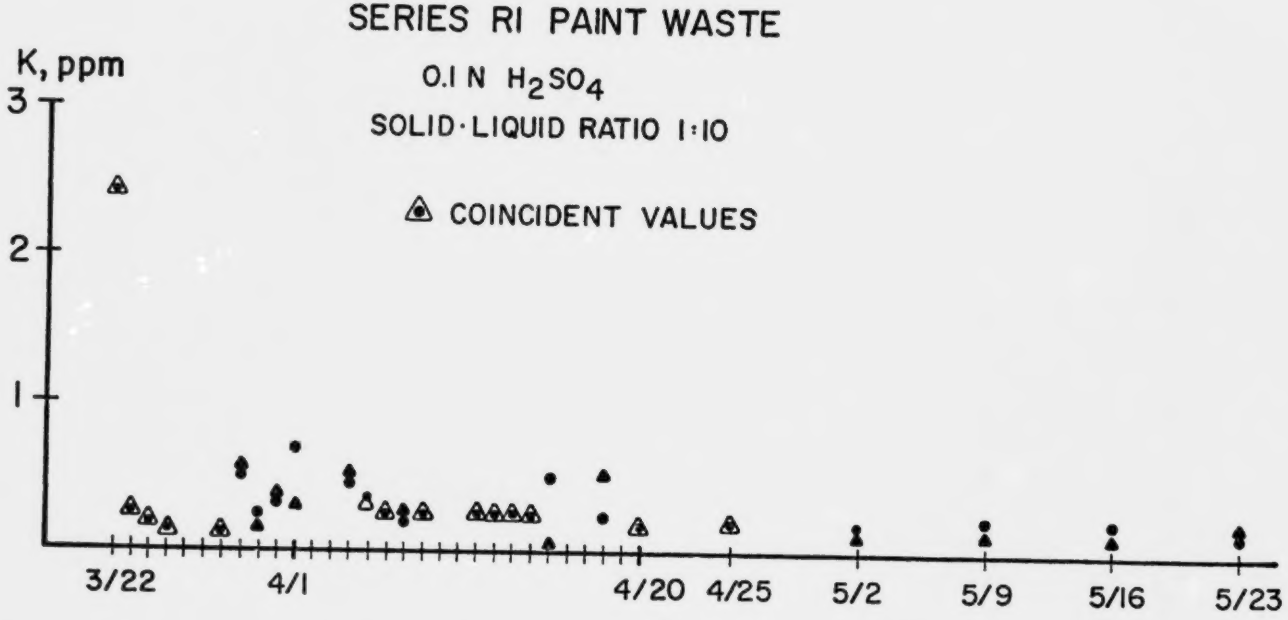

Figure A-80. Expansion of Figure A-79 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

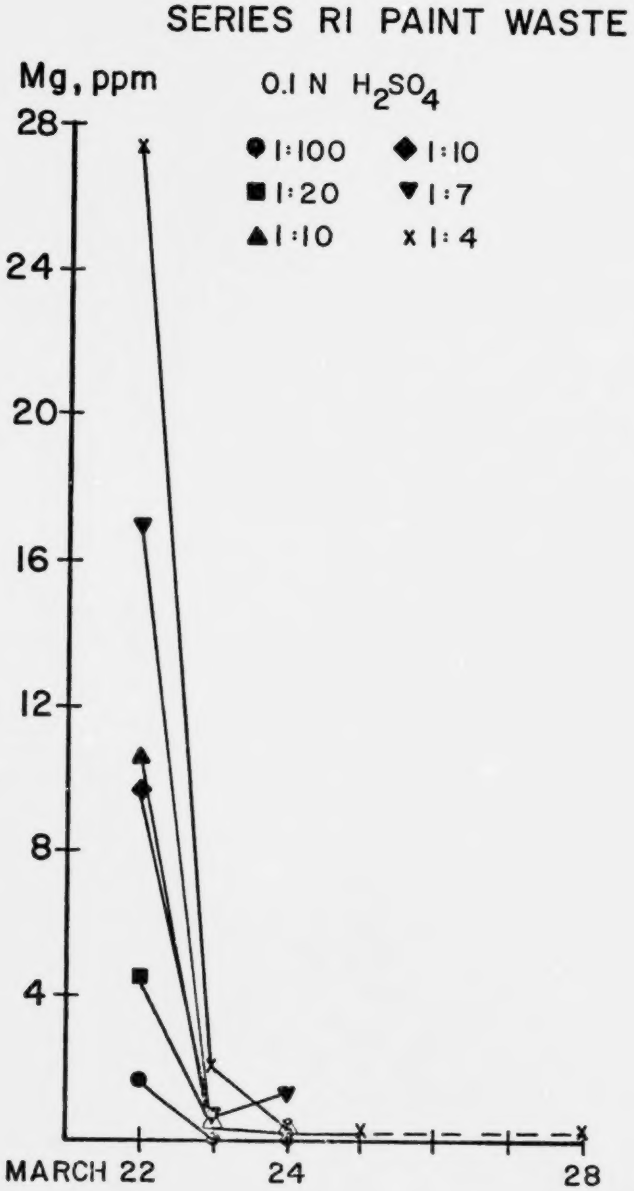

Figure A-81. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. Mg.

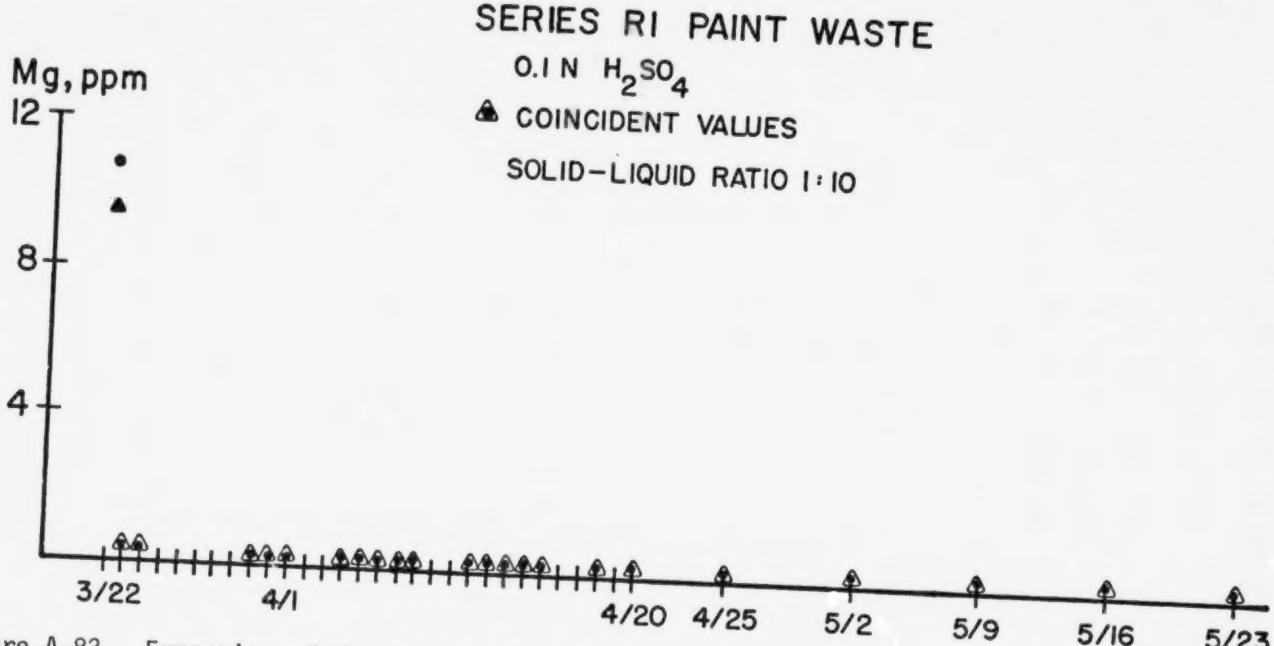

Figure A-82. Expansion of Figure A-81 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

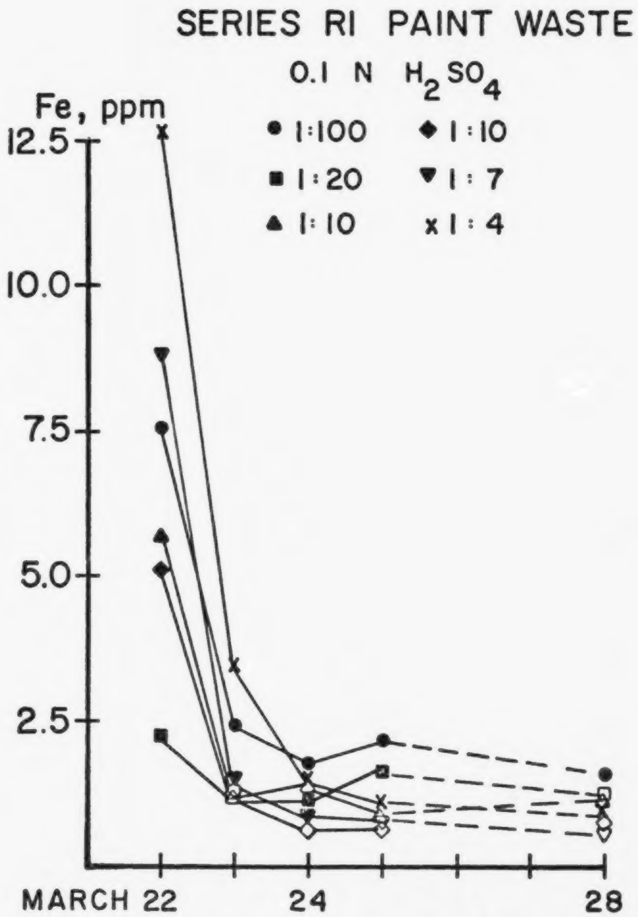

Figure A-83. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. Fe.

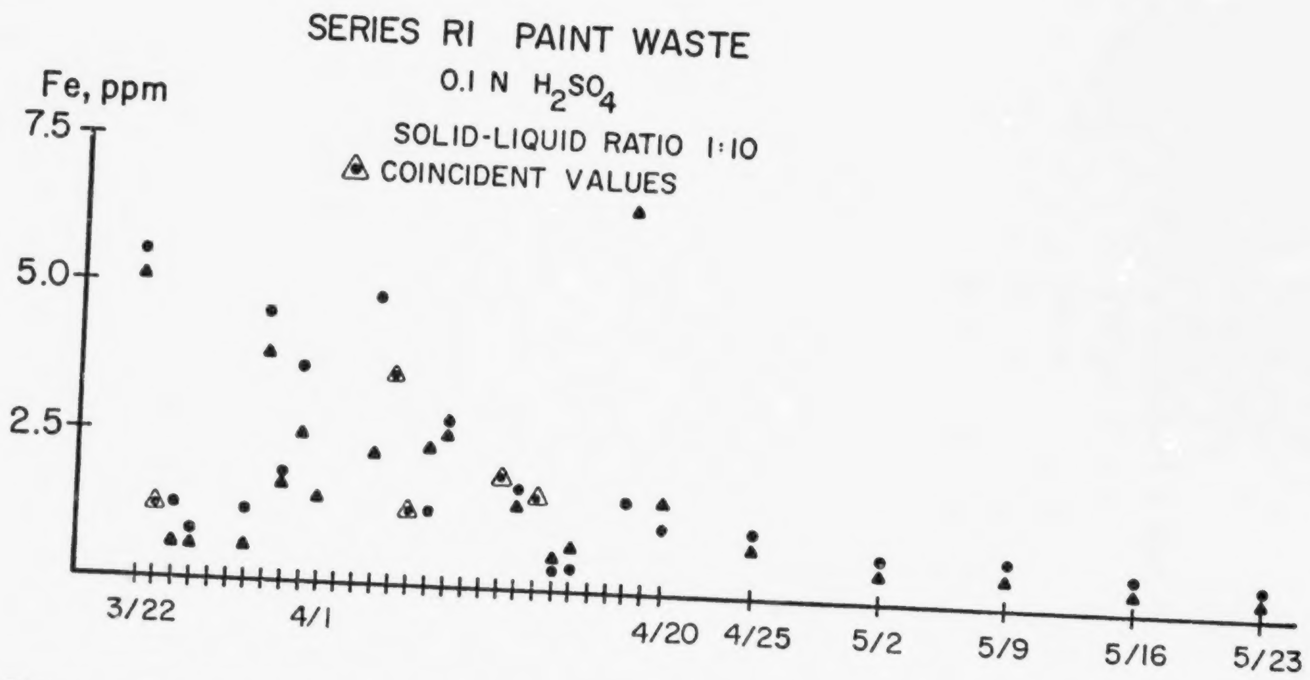

Figure A-84. Expansion of Figure A-83 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

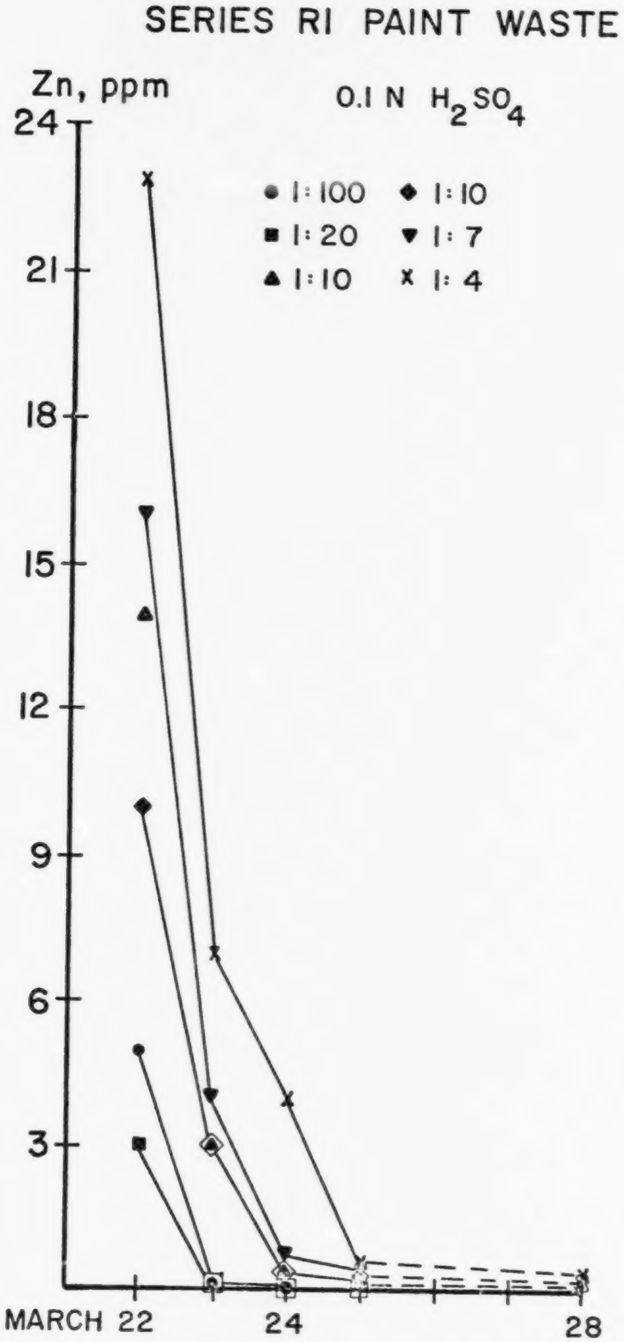

Figure A-85. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. Zn.

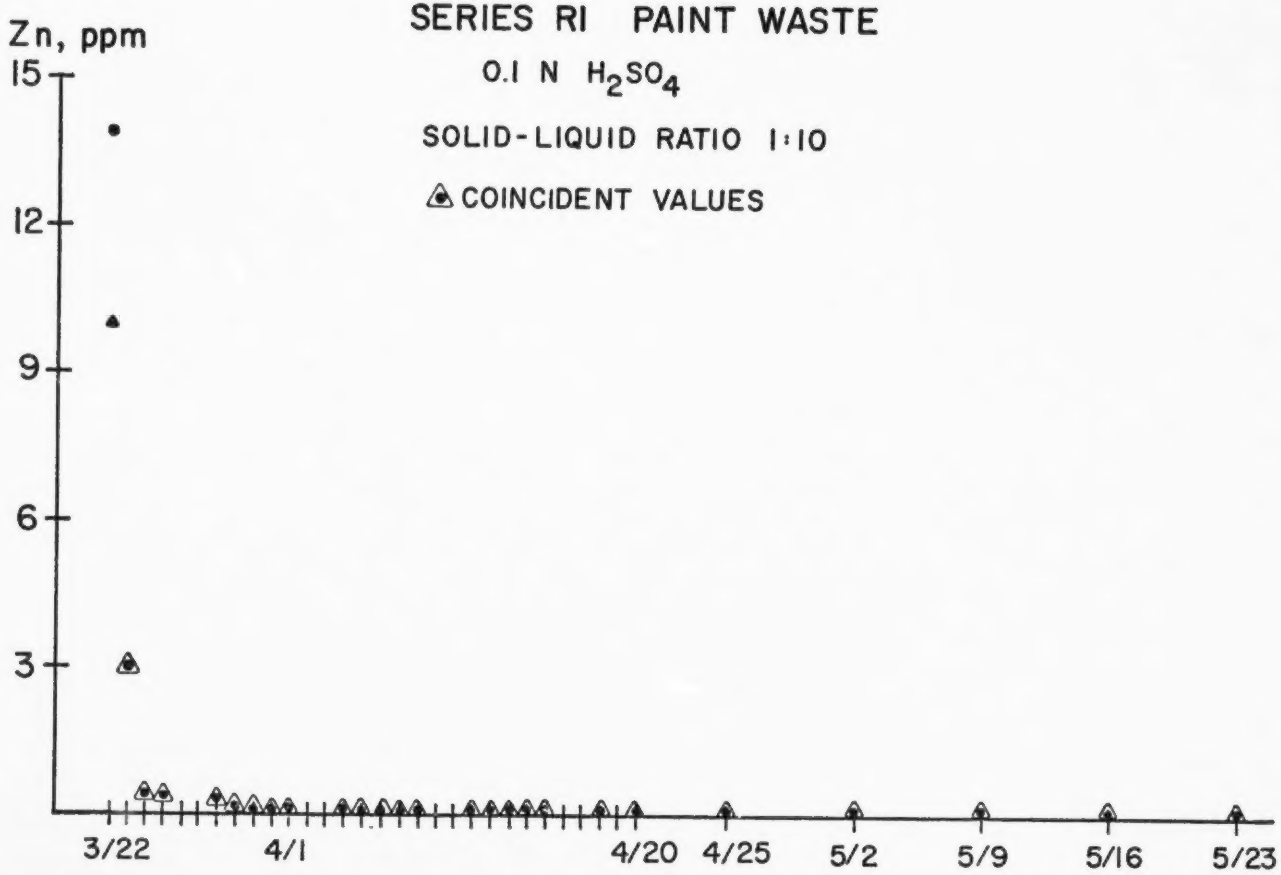

Figure A-86. Expansion of Figure A-85 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

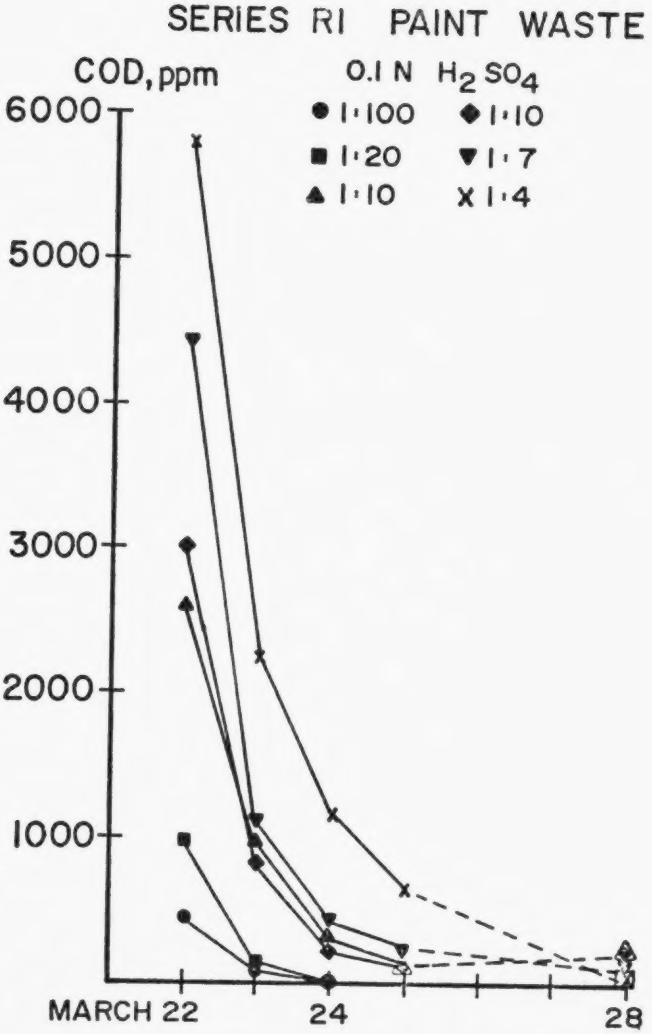

Figure A-87. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. COD.

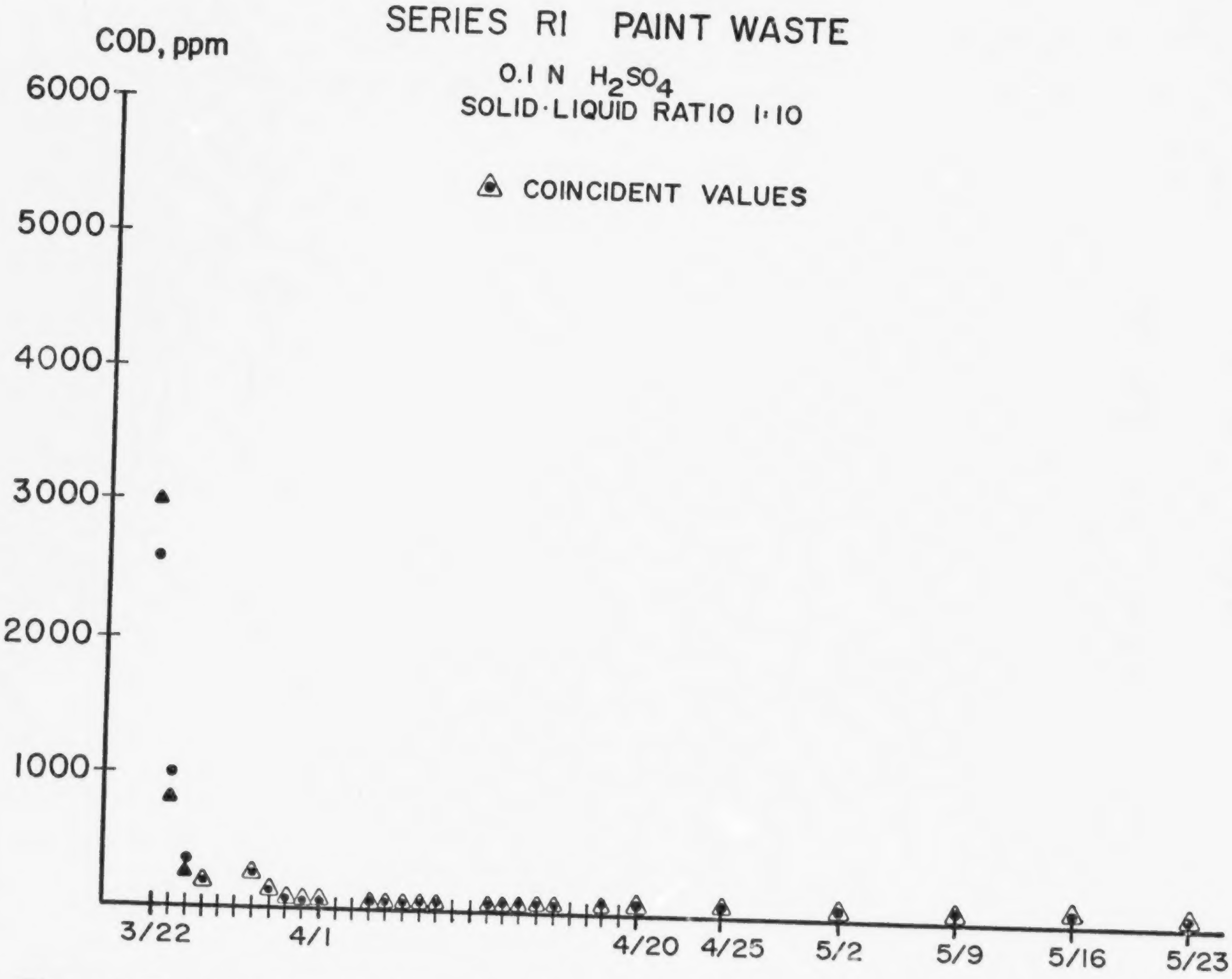

Figure A-88. Expansion of Figure A-87 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

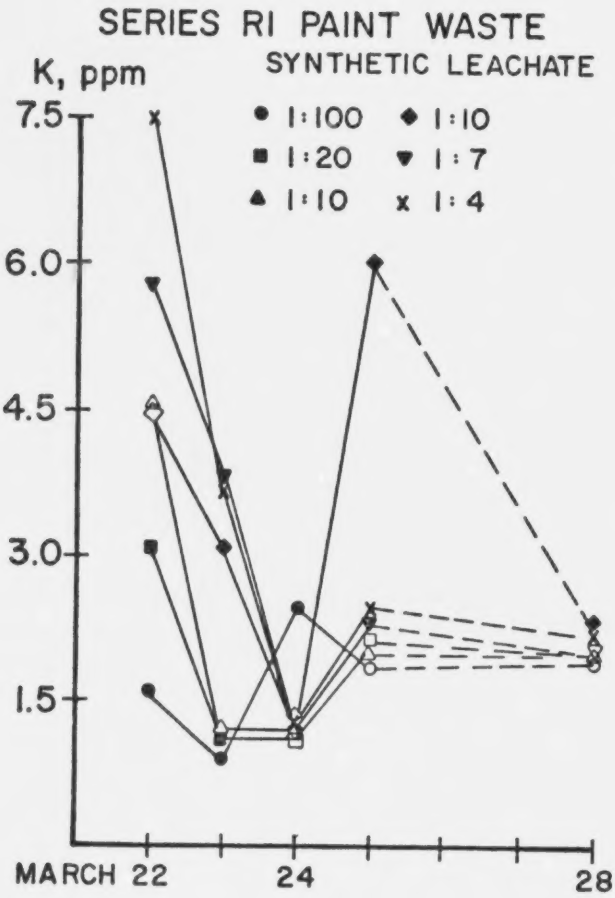

Figure A-89. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and synthetic leachate. K.

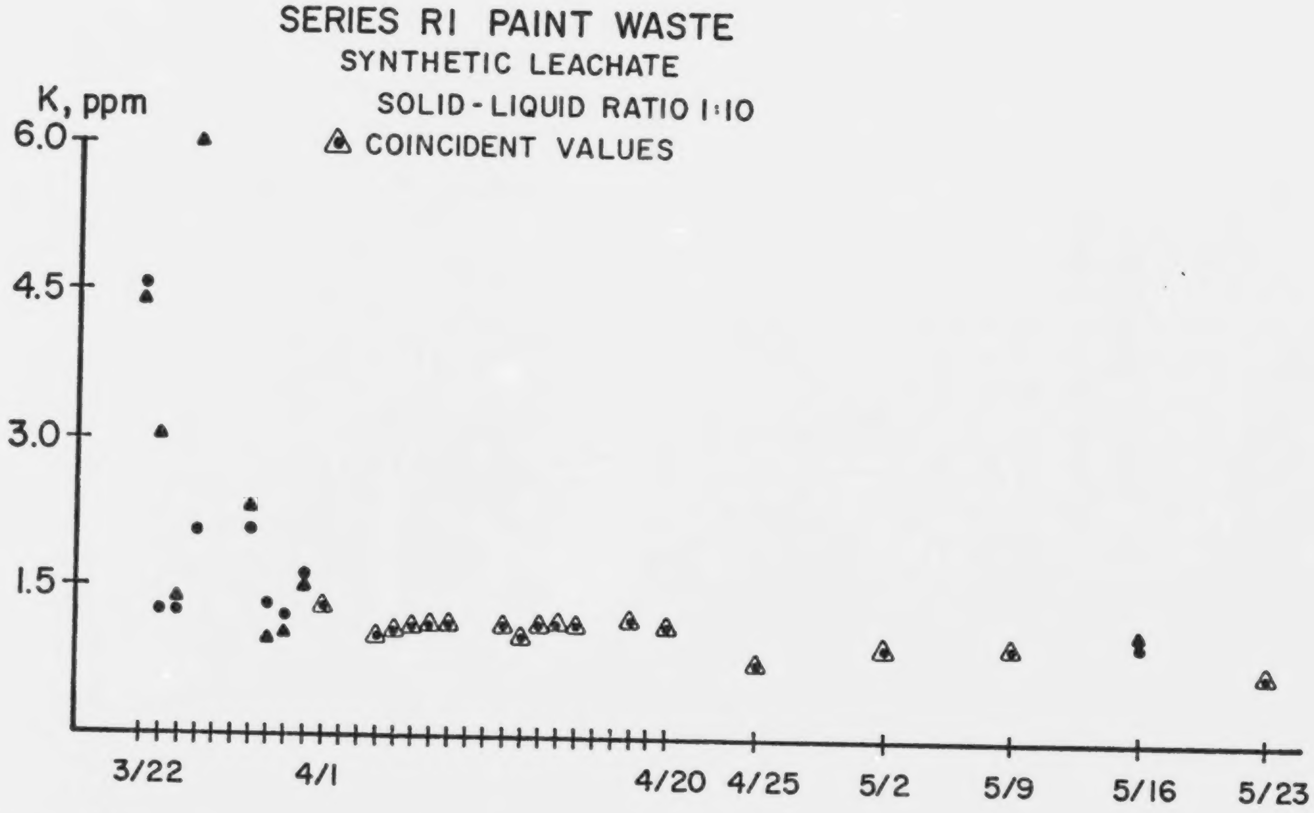

Figure A-90. Expansion of Figure A-89 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

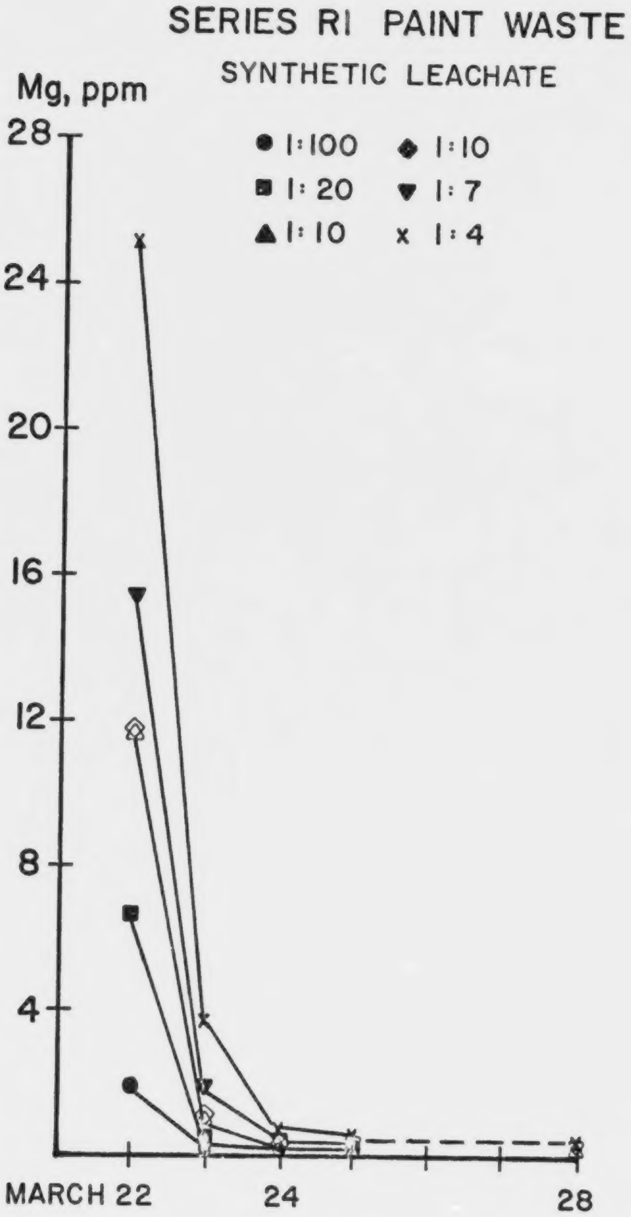

Figure A-91. Test R1 using procedure R to evaluate different solid-liquid ratios over 5 elutions with paint waste and synthetic leachate. Mg.

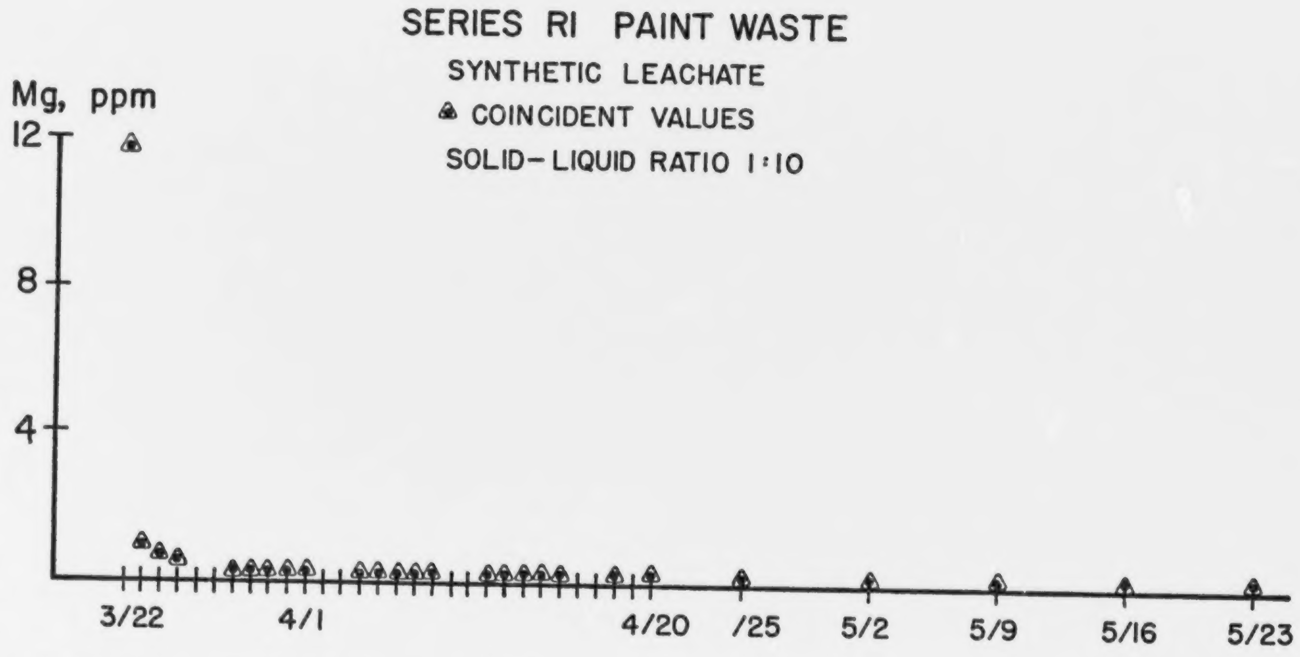

Figure A-92. Expansion of Figure A-91 for a solid-liquid ratio of 1:10 over more elutions. Duplicate runs.

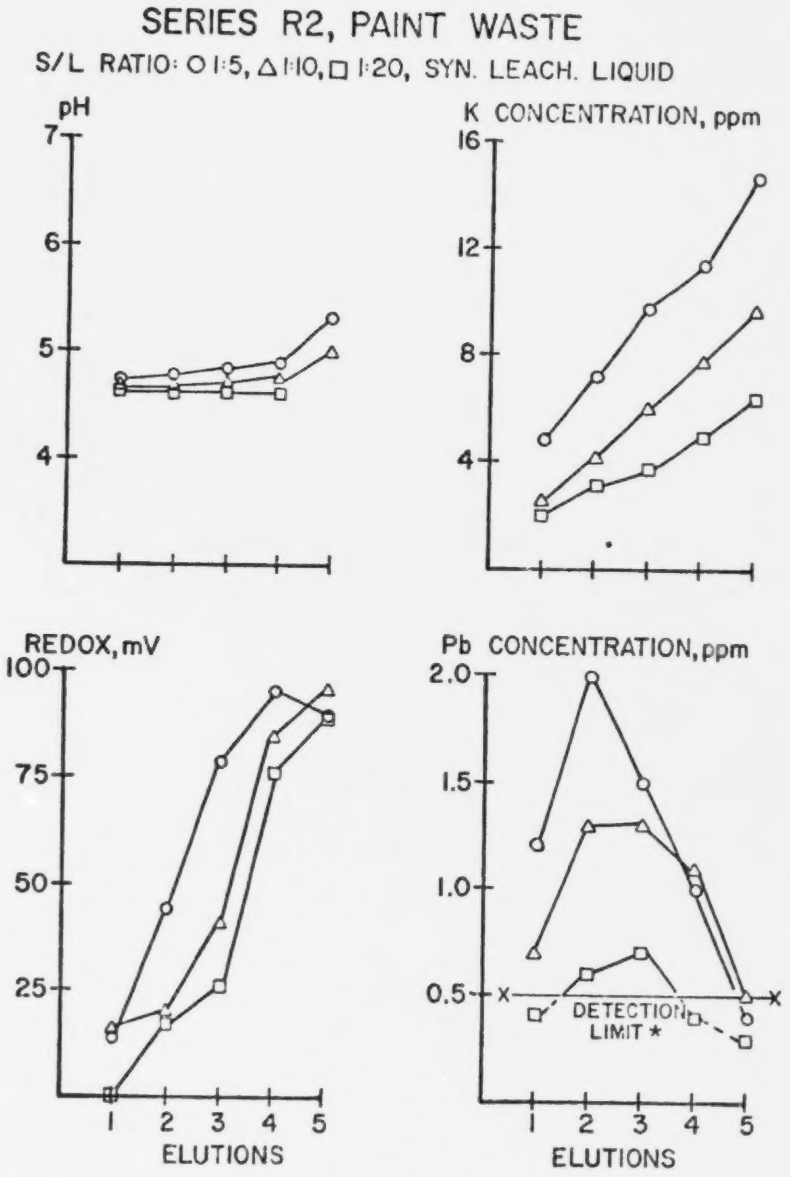

Figure A-93. Test R2 using procedure C to evaluate different solid-liquid ratios over 5 elutions with paint waste and synthetic leachate. pH, K, Redox, Pb.

*All points below detection limit are approximate.

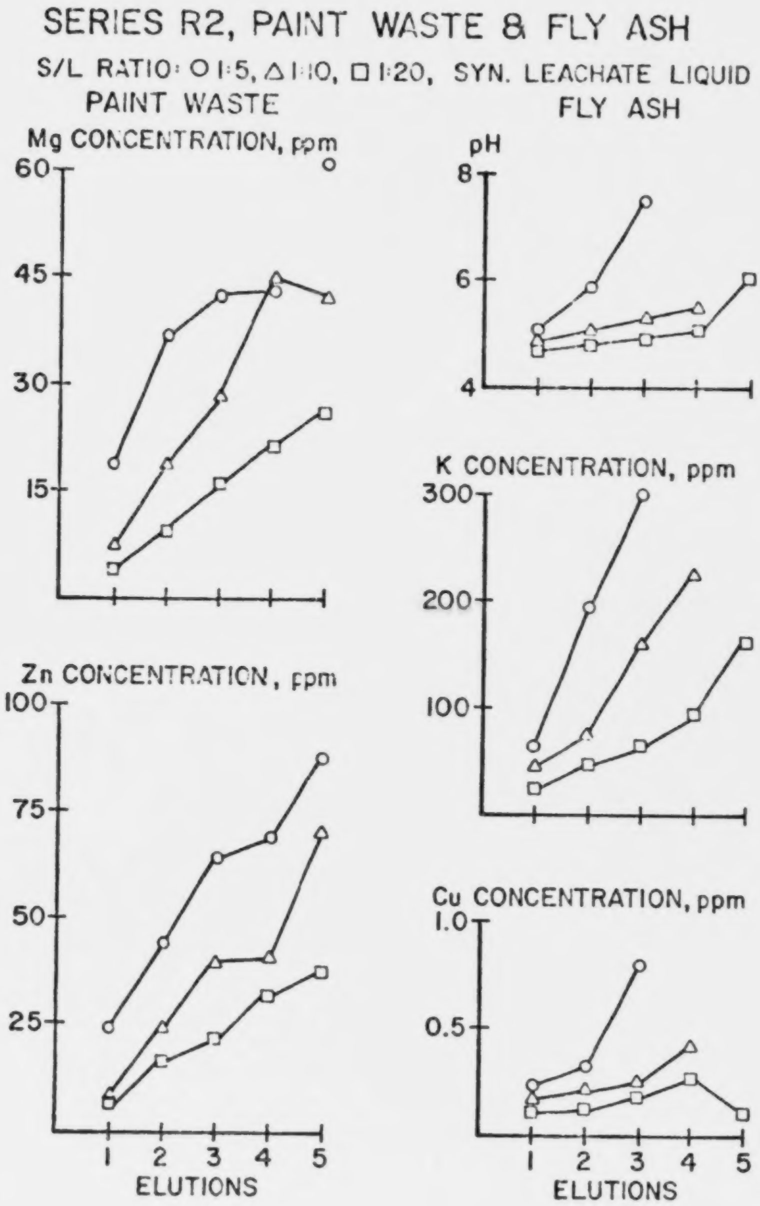

Figure A-94. Test R2 using procedure C to evaluate different solid-liquid ratios over 5 elutions with paint waste and fly ash using synthetic leachate. Mg, Zn, pH, K, Cu.

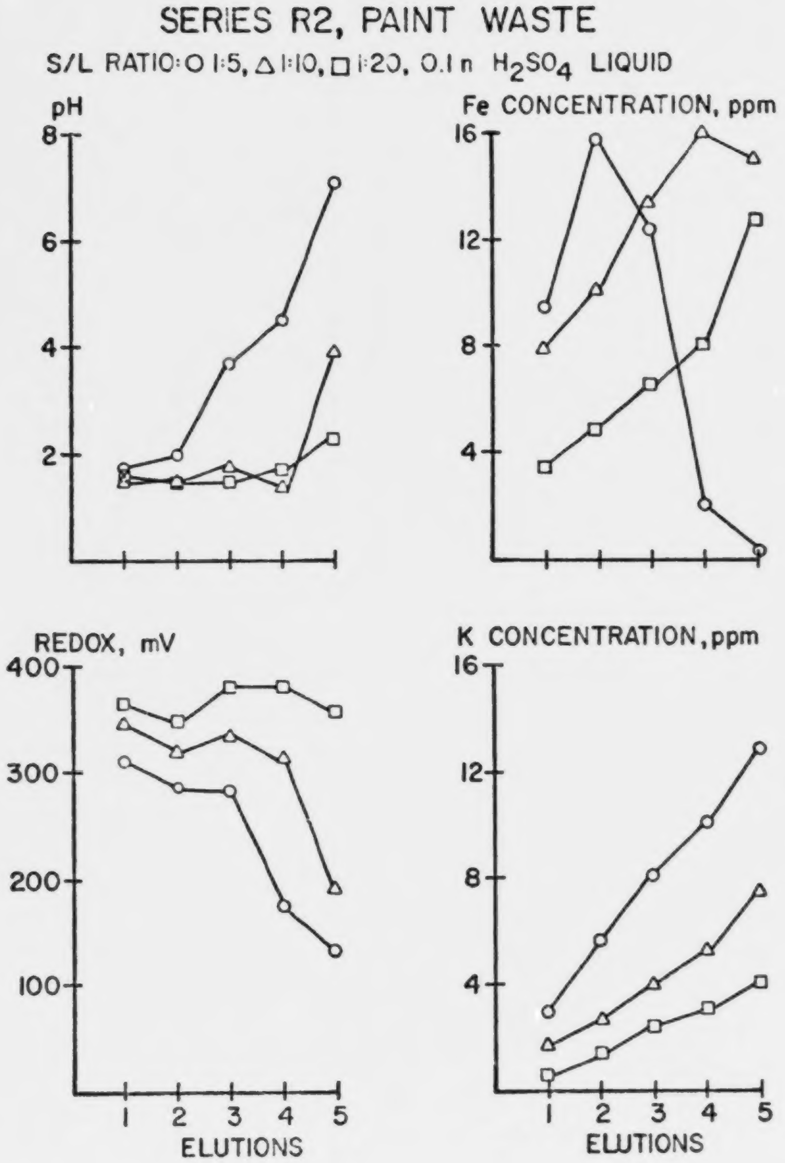

Figure A-95. Test R2 using procedure C to evaluate different solid-liquid ratios over 5 elutions with paint waste and 0.1N H_2SO_4. pH, Fe, Redox, K.

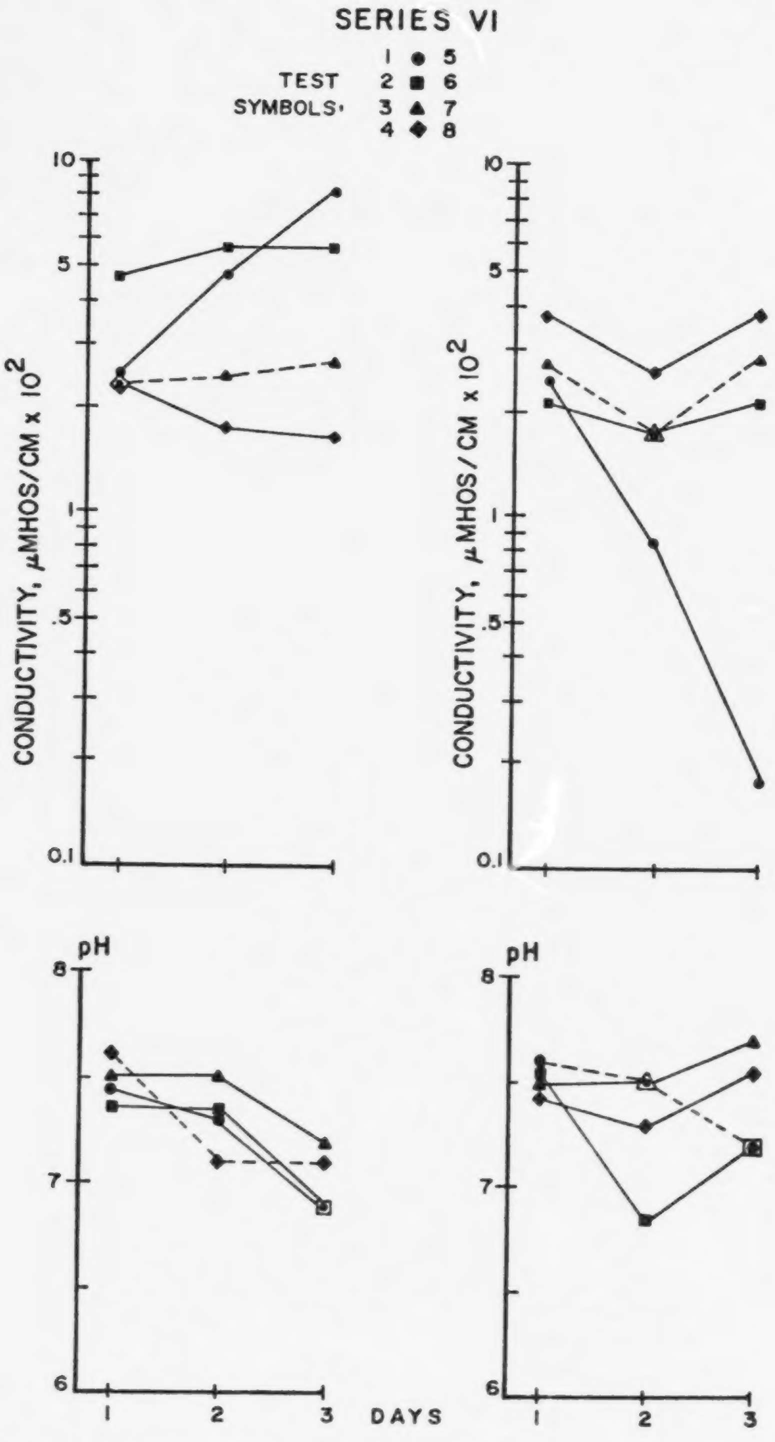

Figure A-96. Test VI evaluating different contact procedures with fly ash (EPA) and distilled water. (See text for procedure.) Specific conductance and pH.

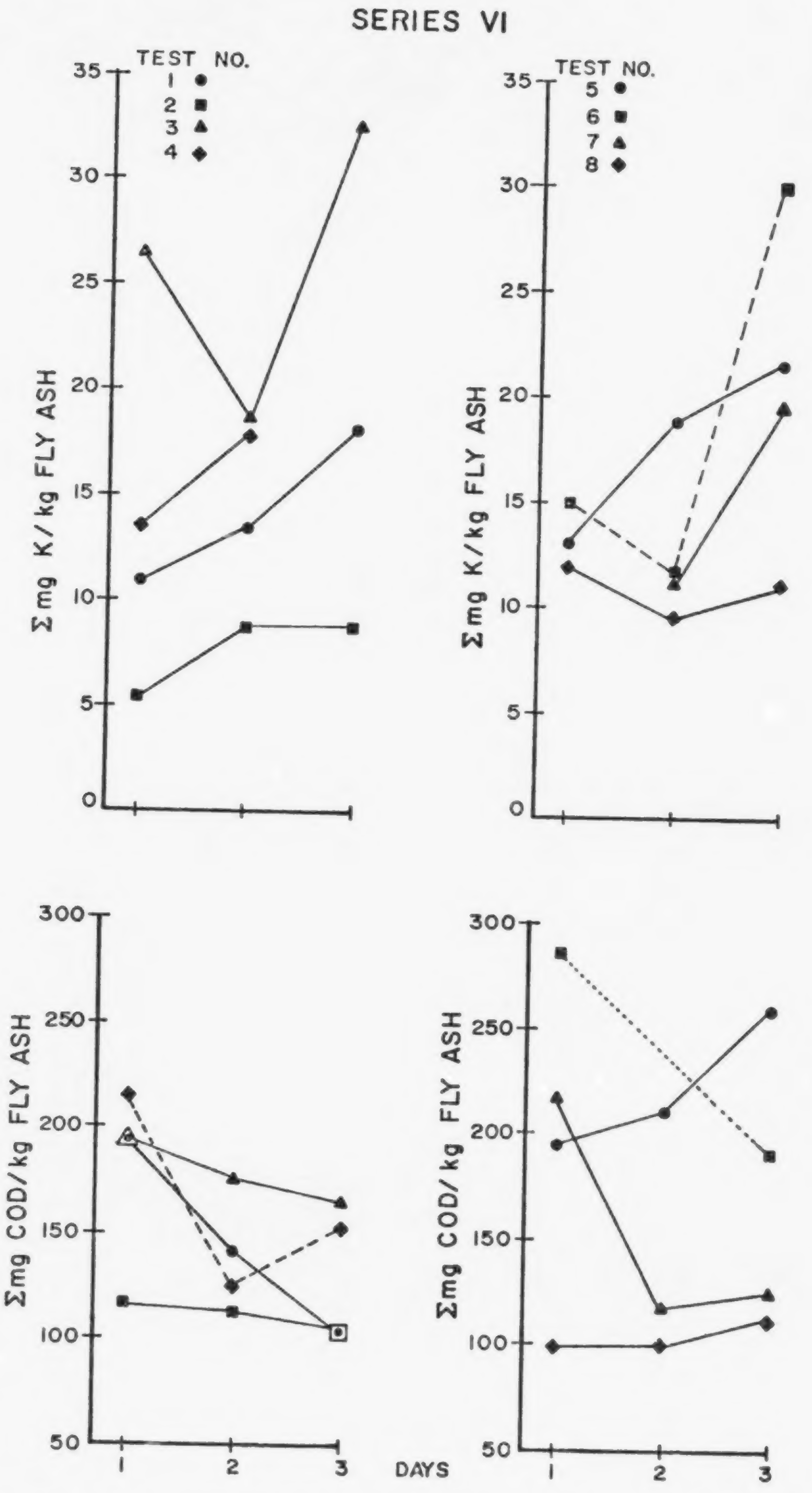

Figure A-97. Test VI evaluating different contact procedures with fly ash (EPA) and distilled water. K and COD.

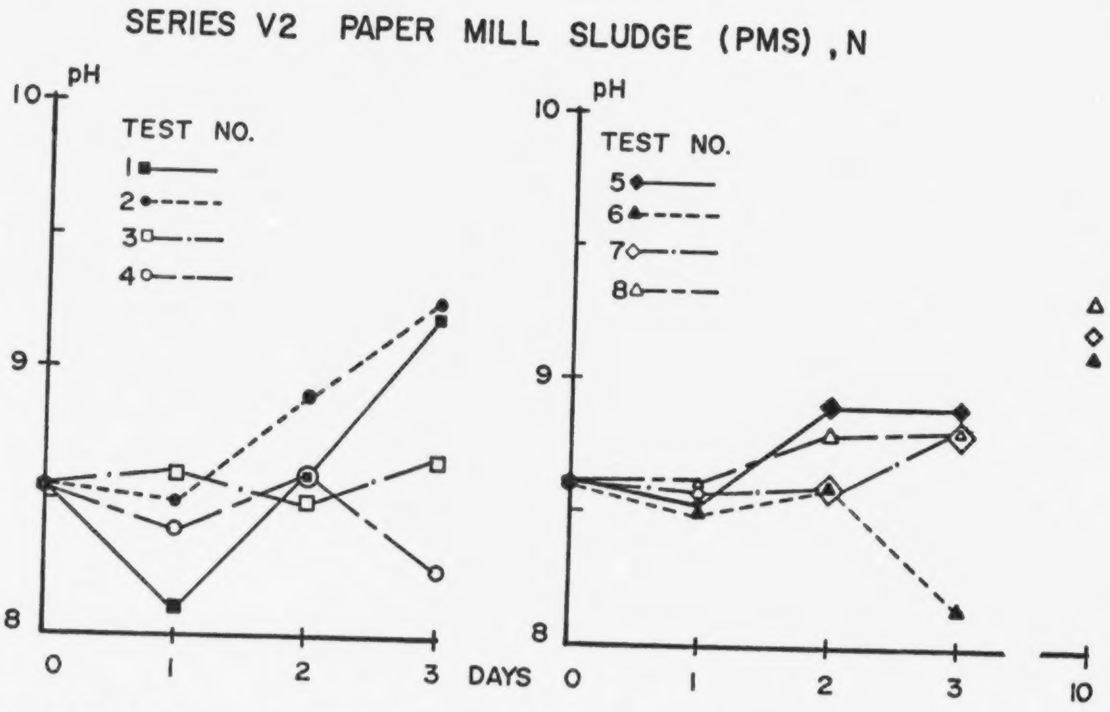

Figure A-98. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. (See text for procedure.) pH.

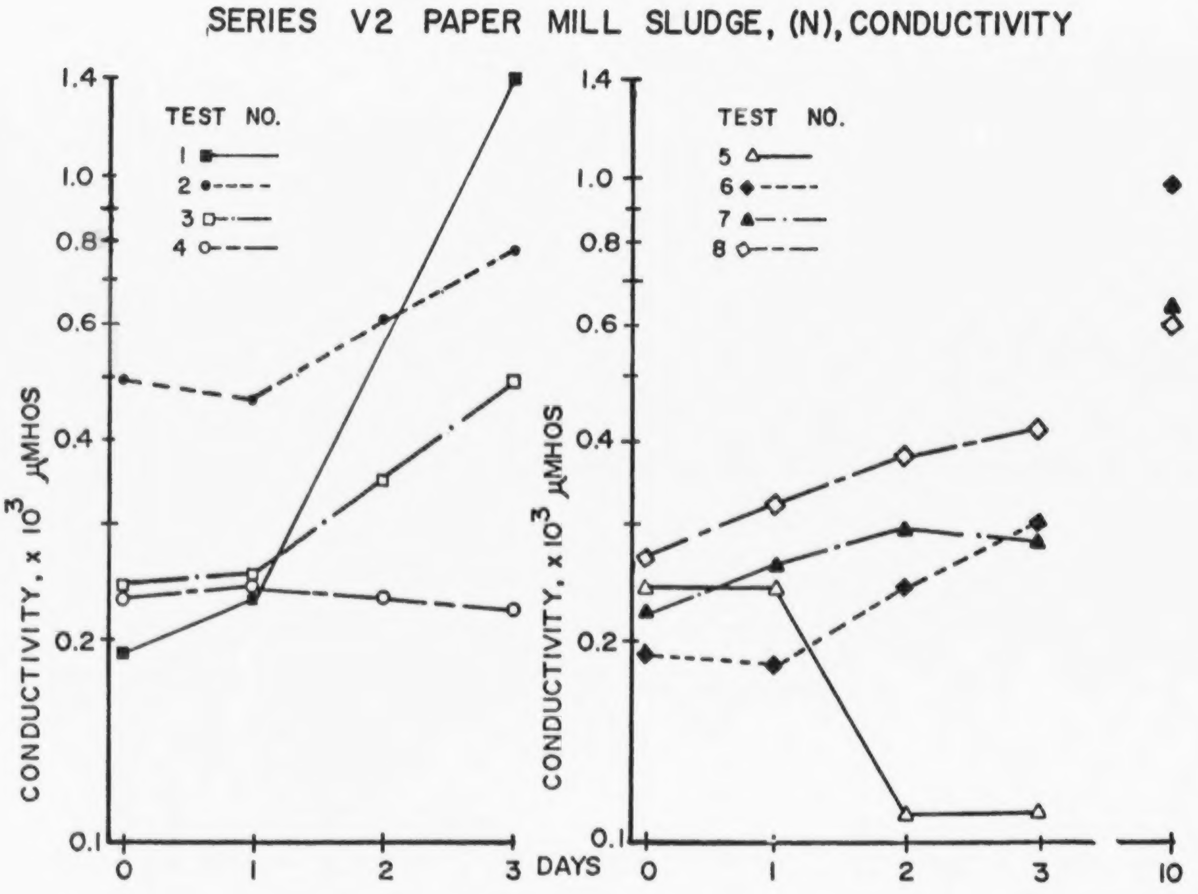

Figure A-99. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. Specific conductance.

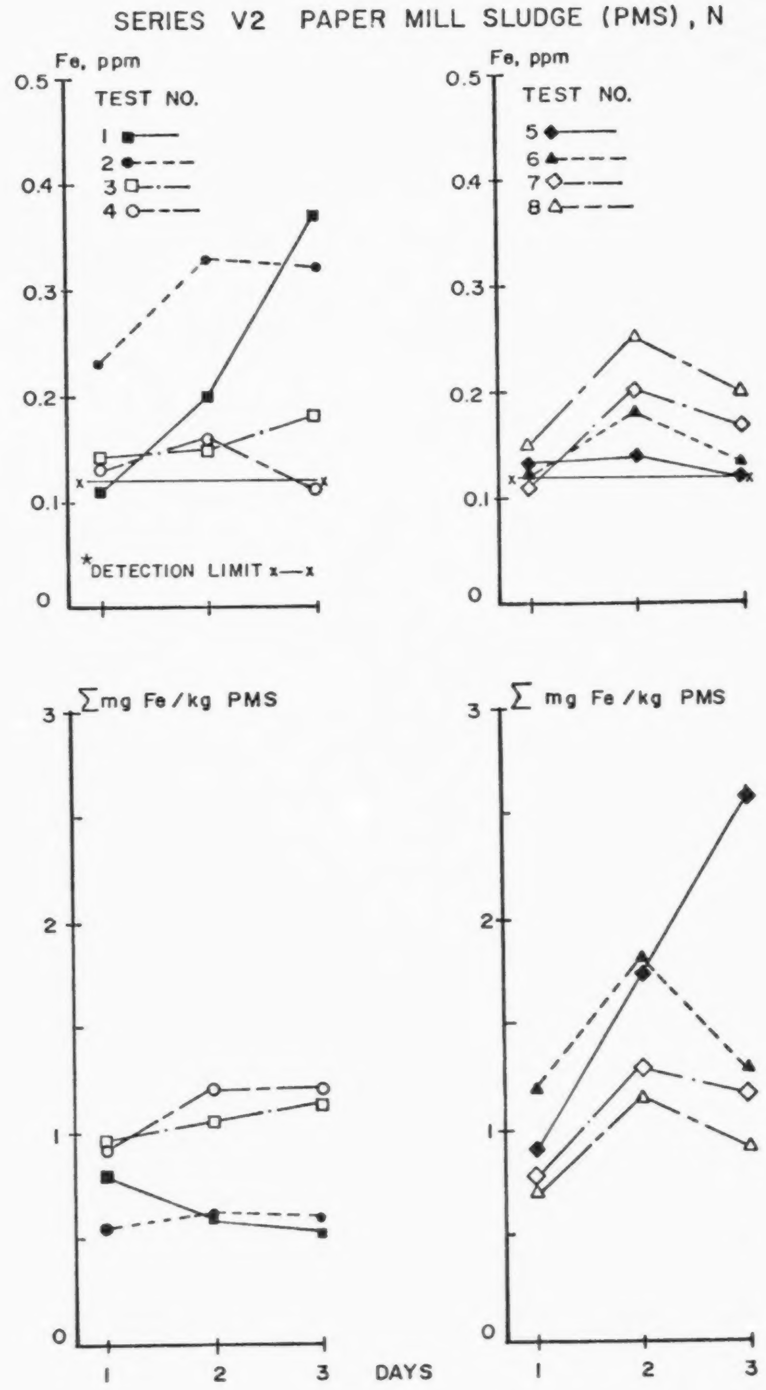

Figure A-100. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. Fe.

*All points below detection limit are approximate.

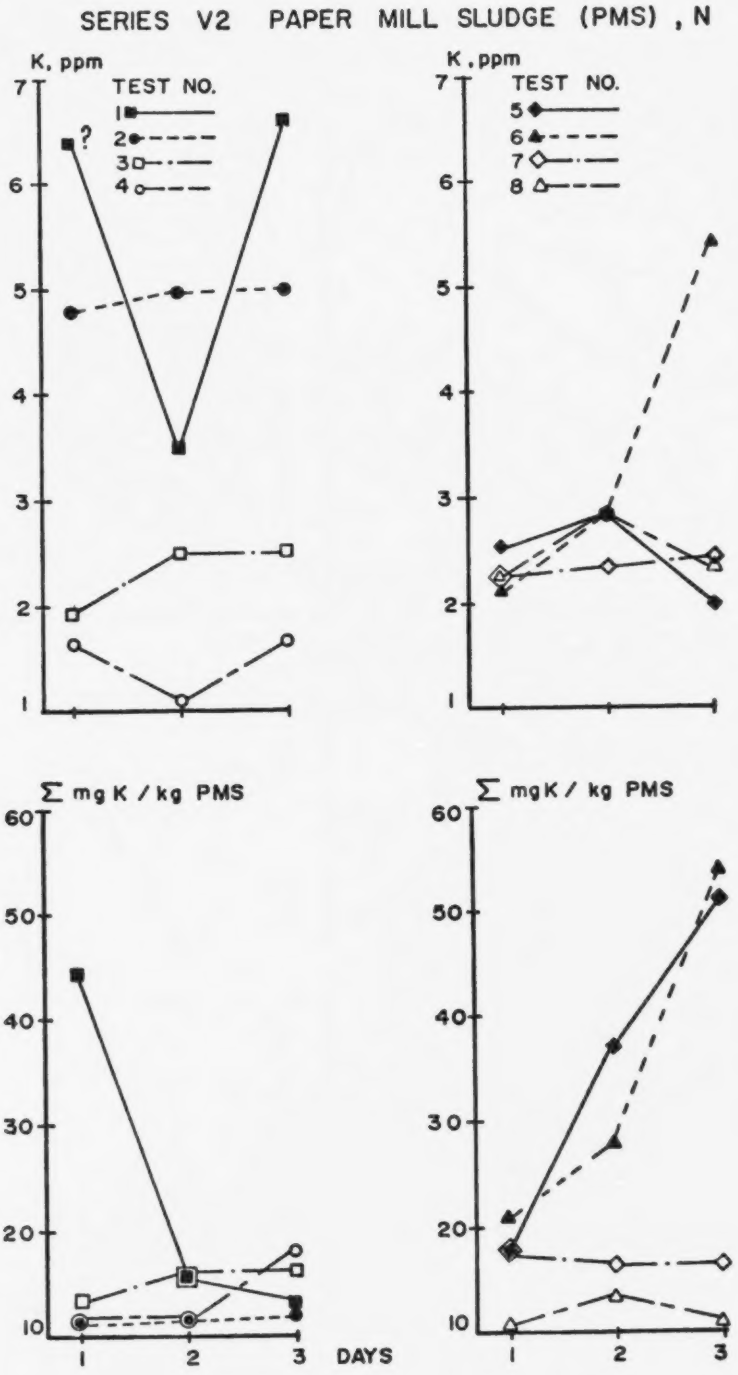

Figure A-101. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. K.

SERIES V2 PAPER MILL SLUDGE (PMS), N

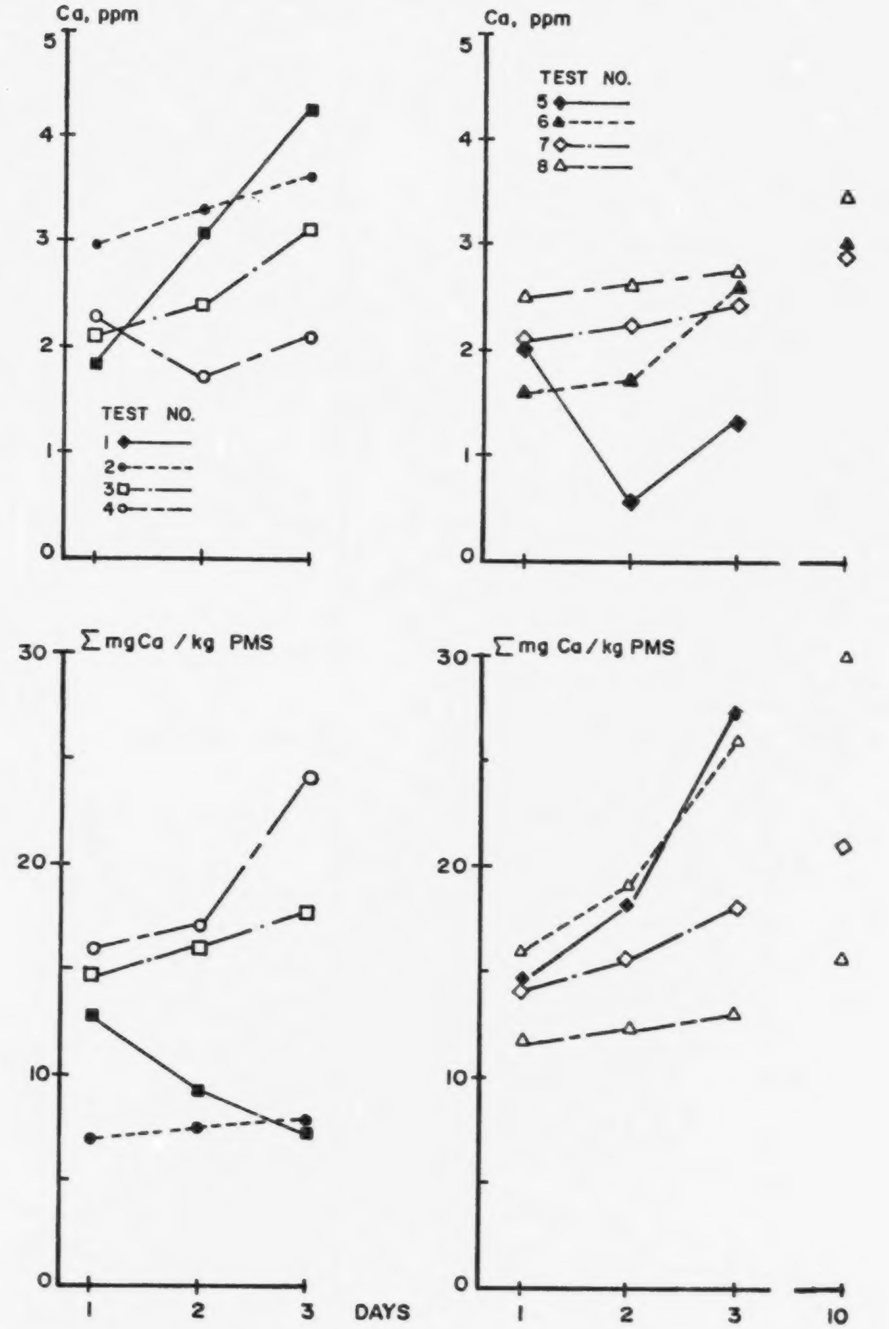

Figure A-102. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. Ca.

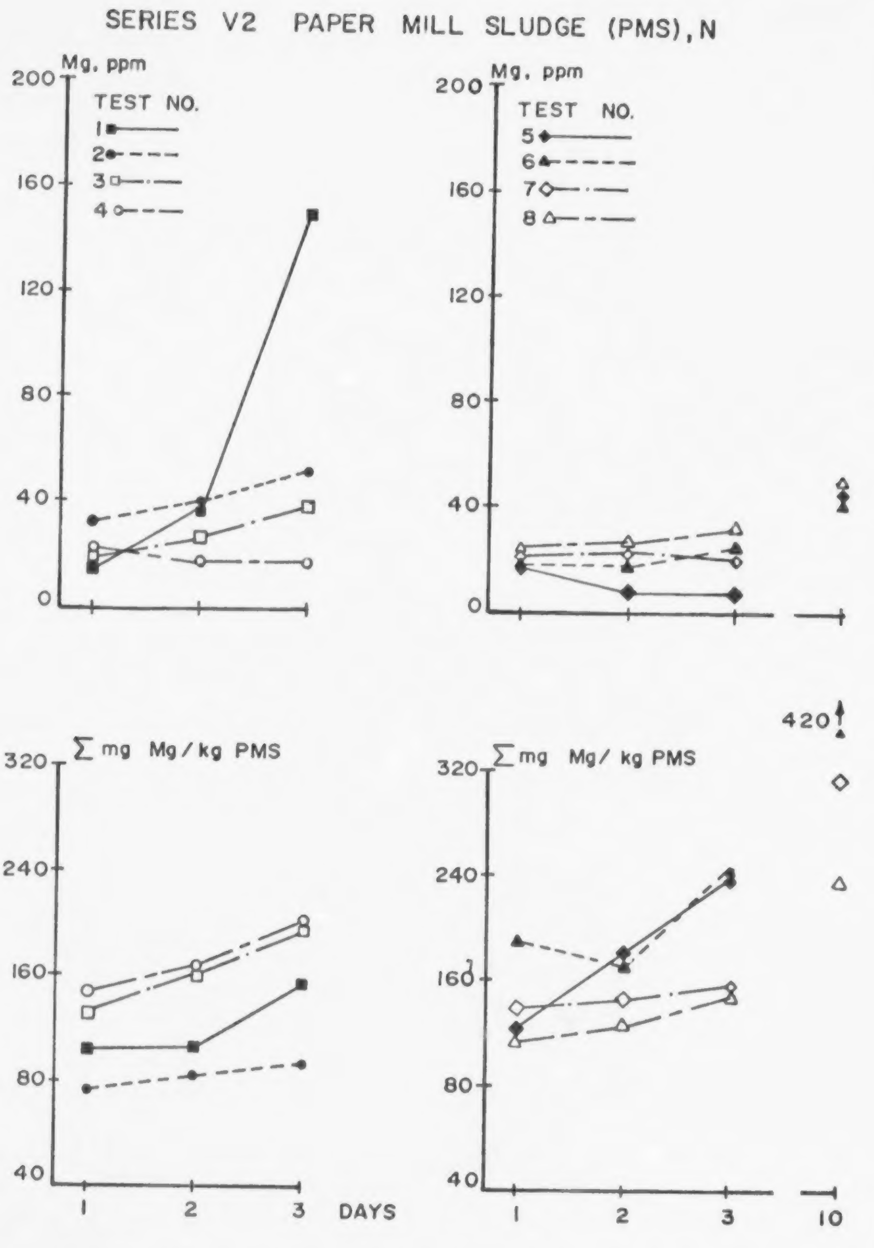

Figure A-103. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. Mg.

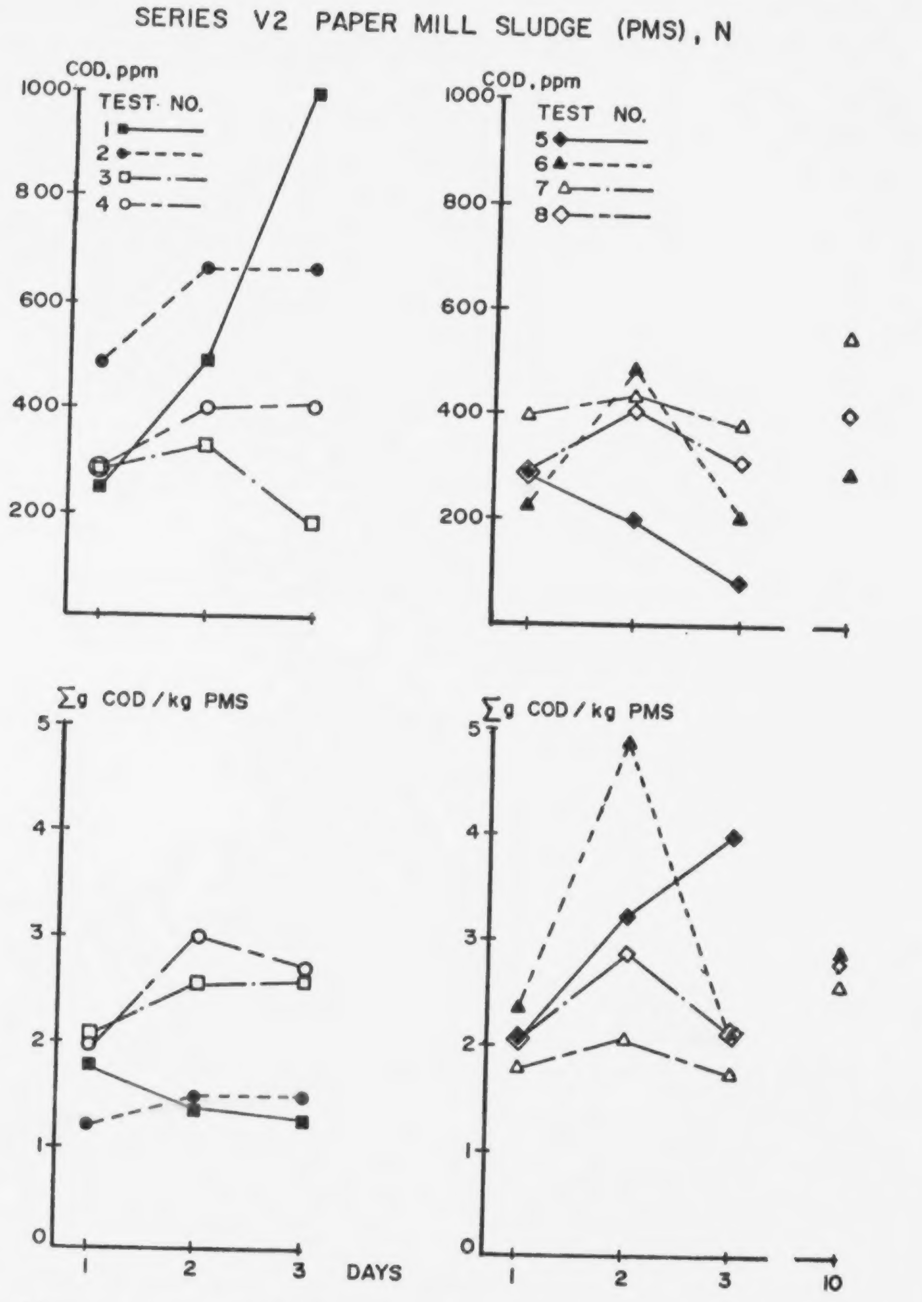

Figure A-104. Test V2 evaluating different contact procedures with papermill sludge (N) and distilled water. (COD).

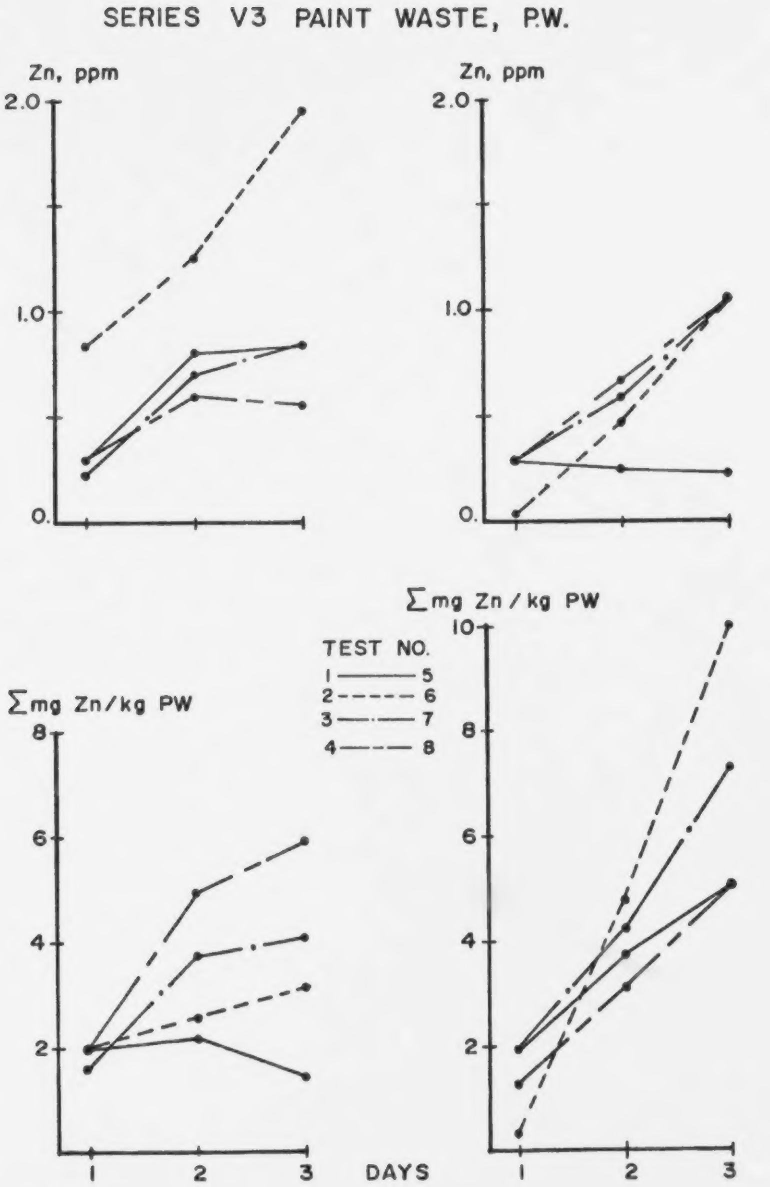

Figure A-105. Test V3 evaluating different contact procedures with paint waste and distilled water. (See text for procedure.) Zn.

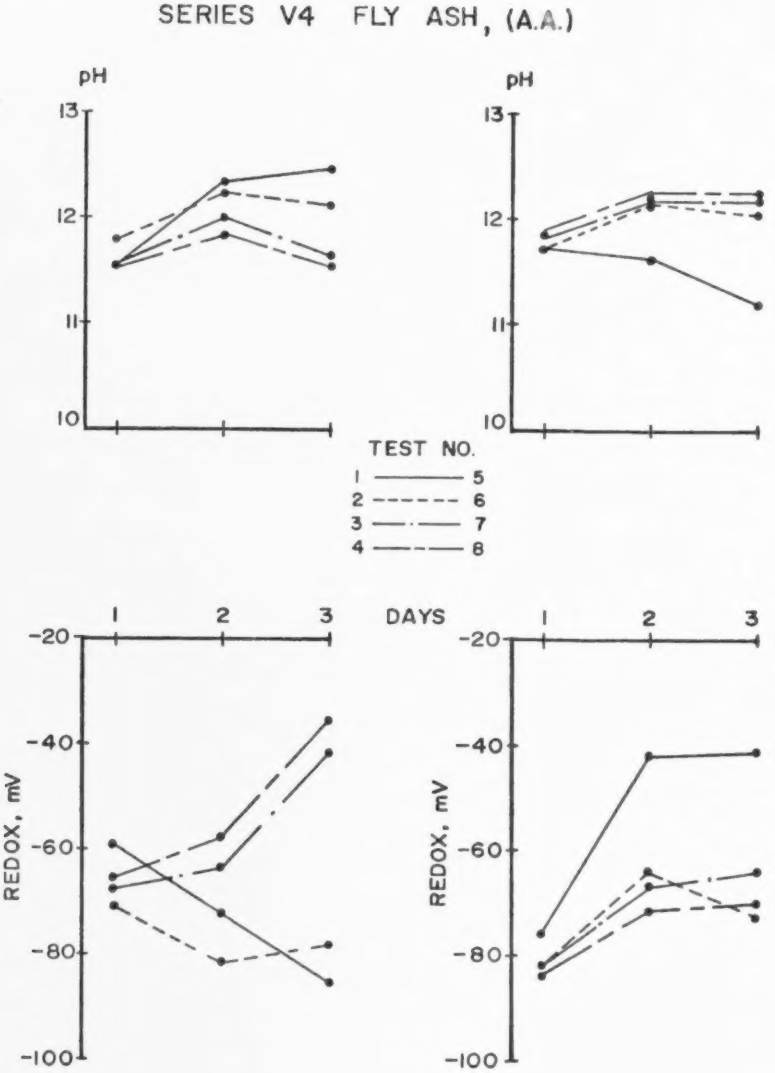

Figure A-106. Test V4 evaluating different contact procedures with fly ash and distilled water. (See text for procedure.) pH and Redox.

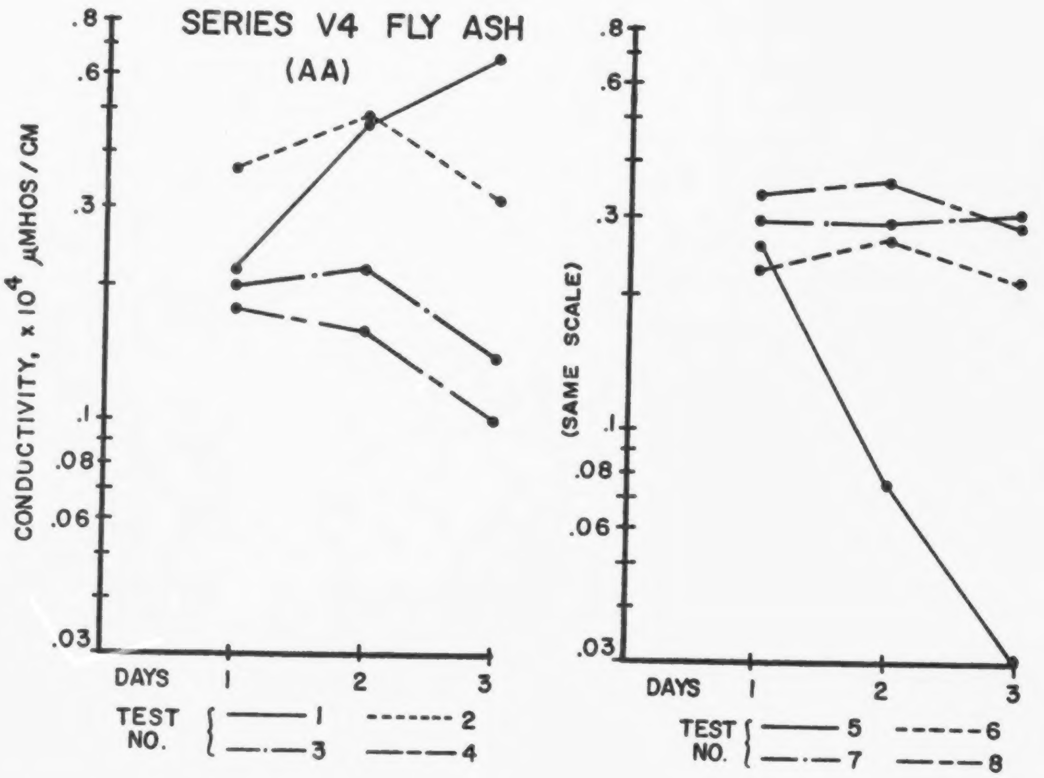

Figure A-107. Test V4 evaluating different contact procedures with fly ash and distilled water. Specific conductance.

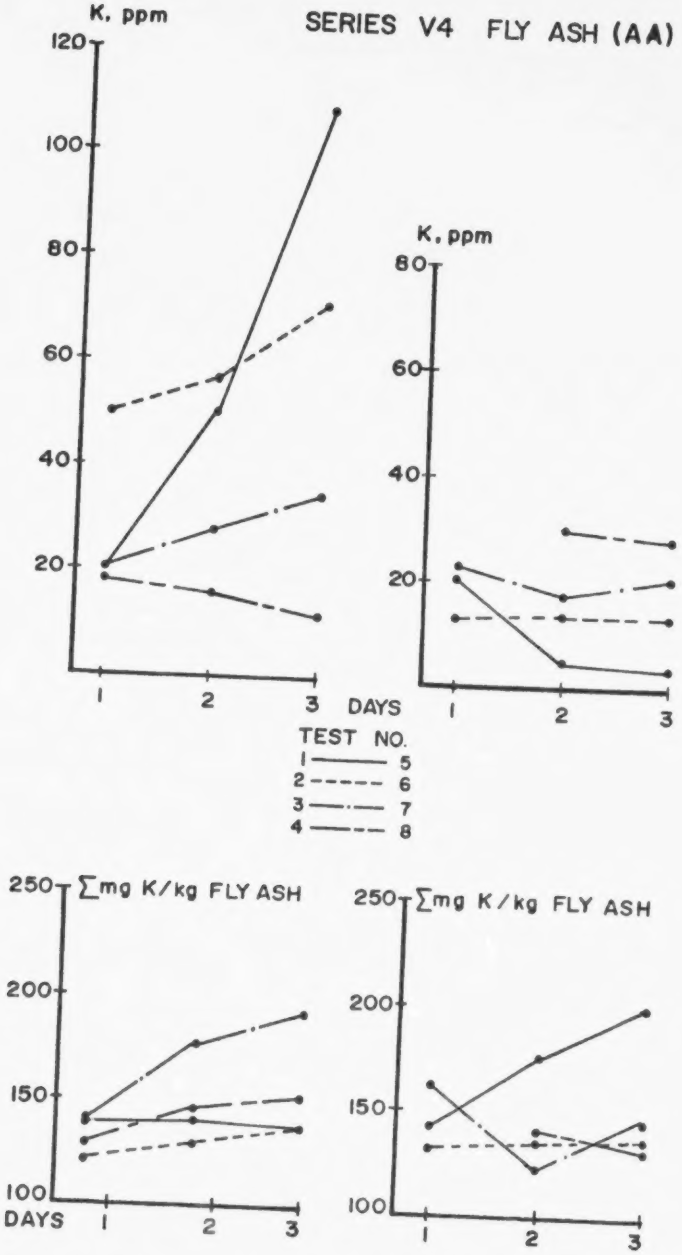

Table A-108. Test V4 evaluating different contact procedures with fly ash and distilled water. K.

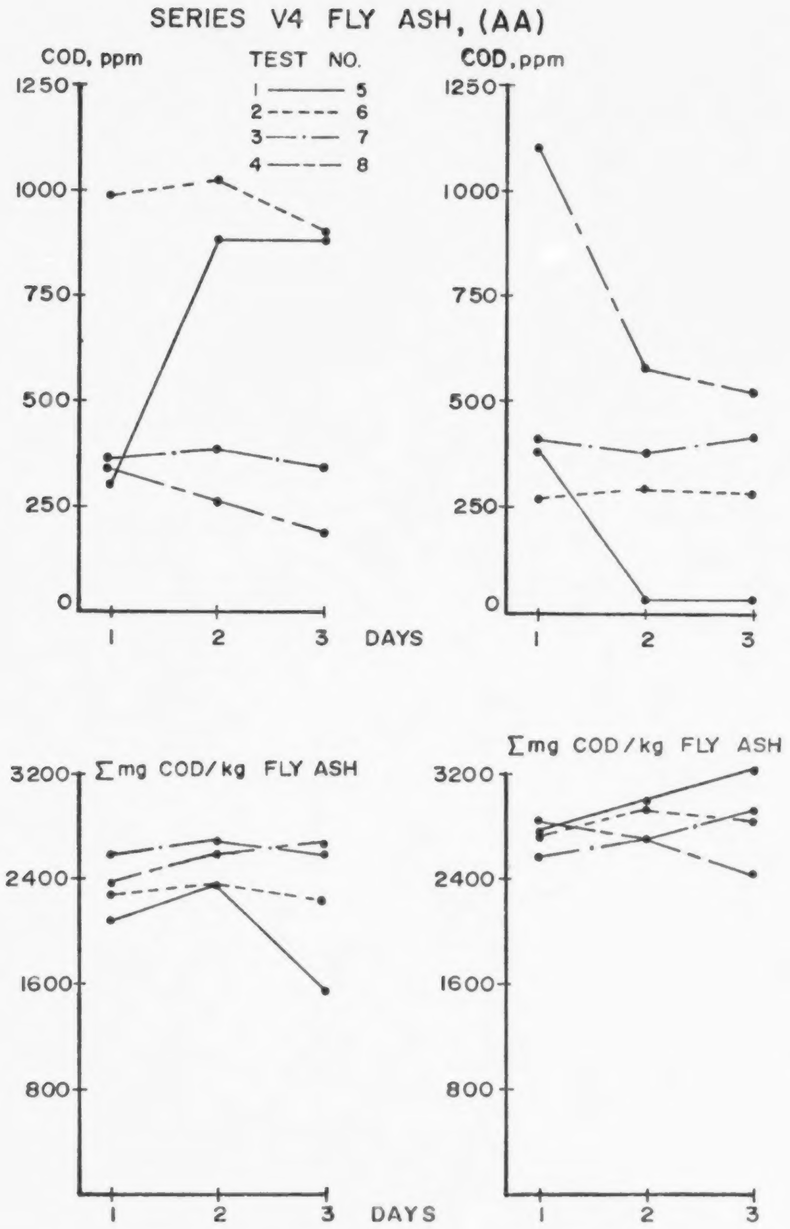

Figure A-109. Test V4 evaluating different contact procedures with fly ash and distilled water. COD.

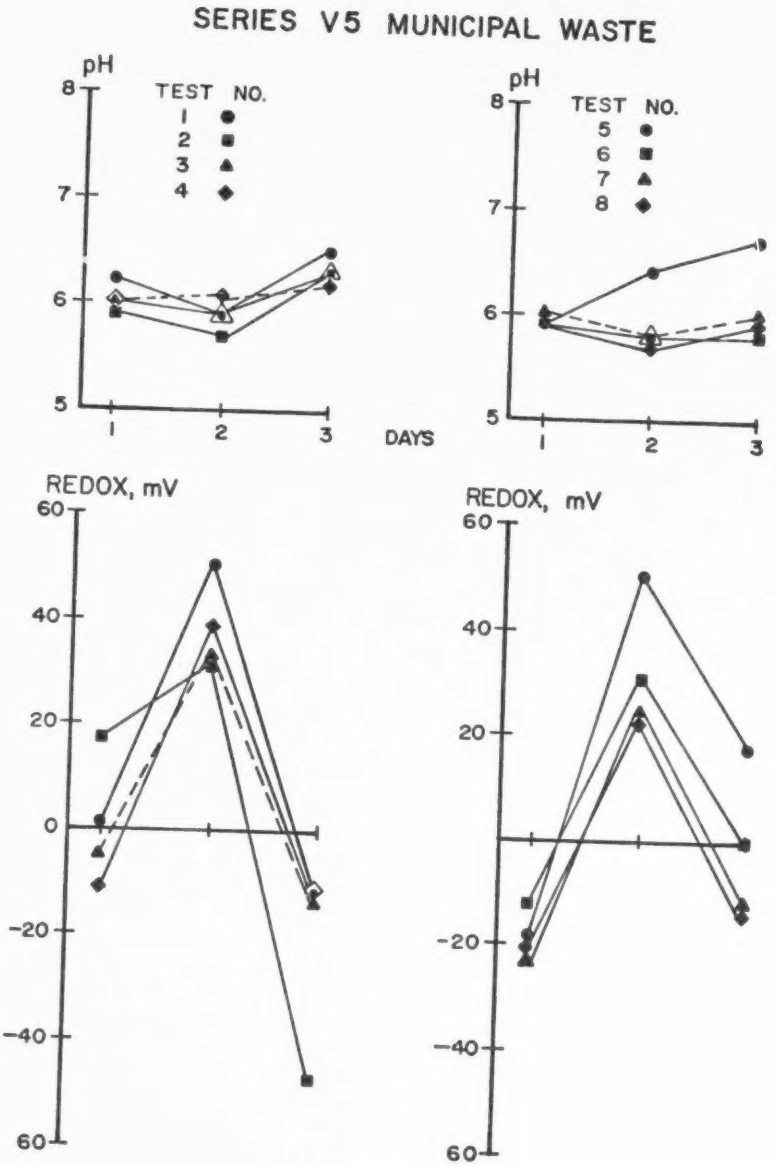

Figure A-110. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. (See text for procedure.) pH and Redox.

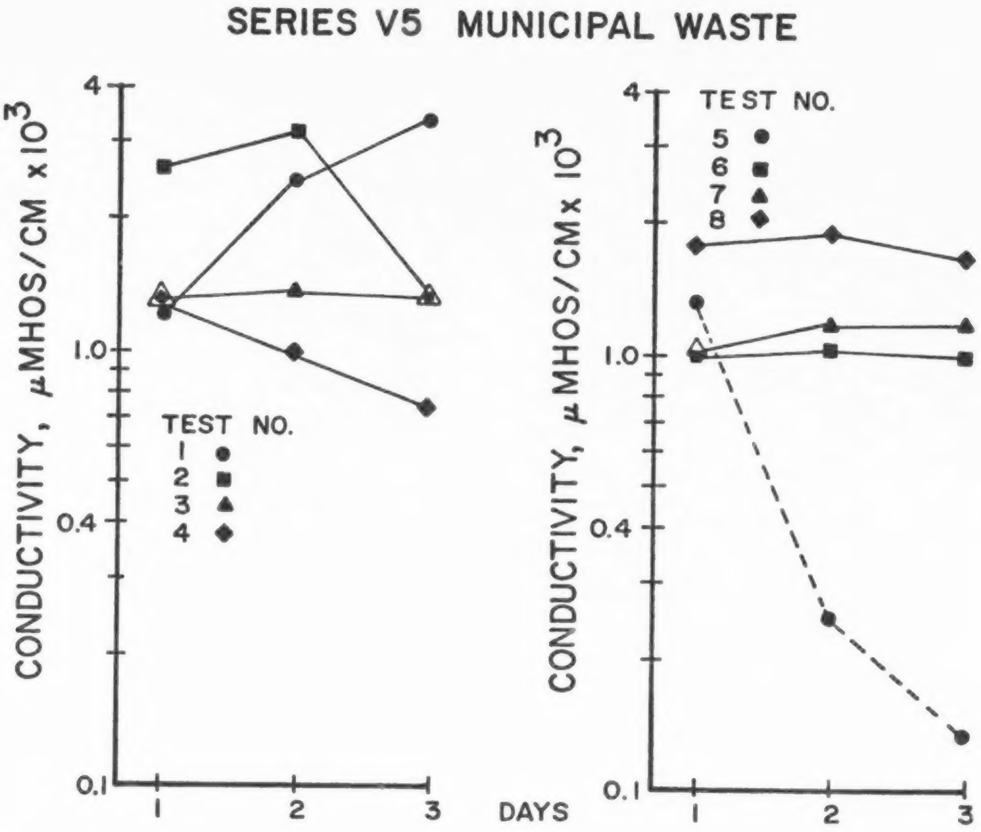

Figure A-111. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Specific conductance.

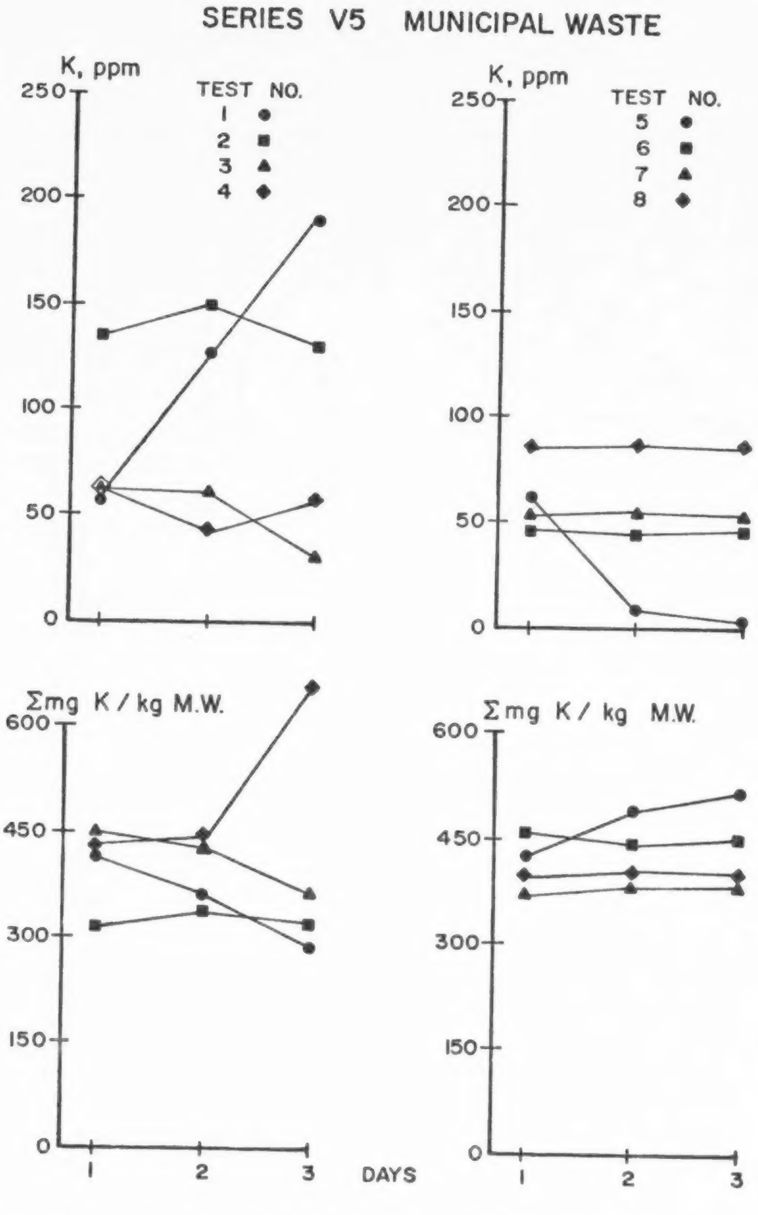

Figure A-112. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. K.

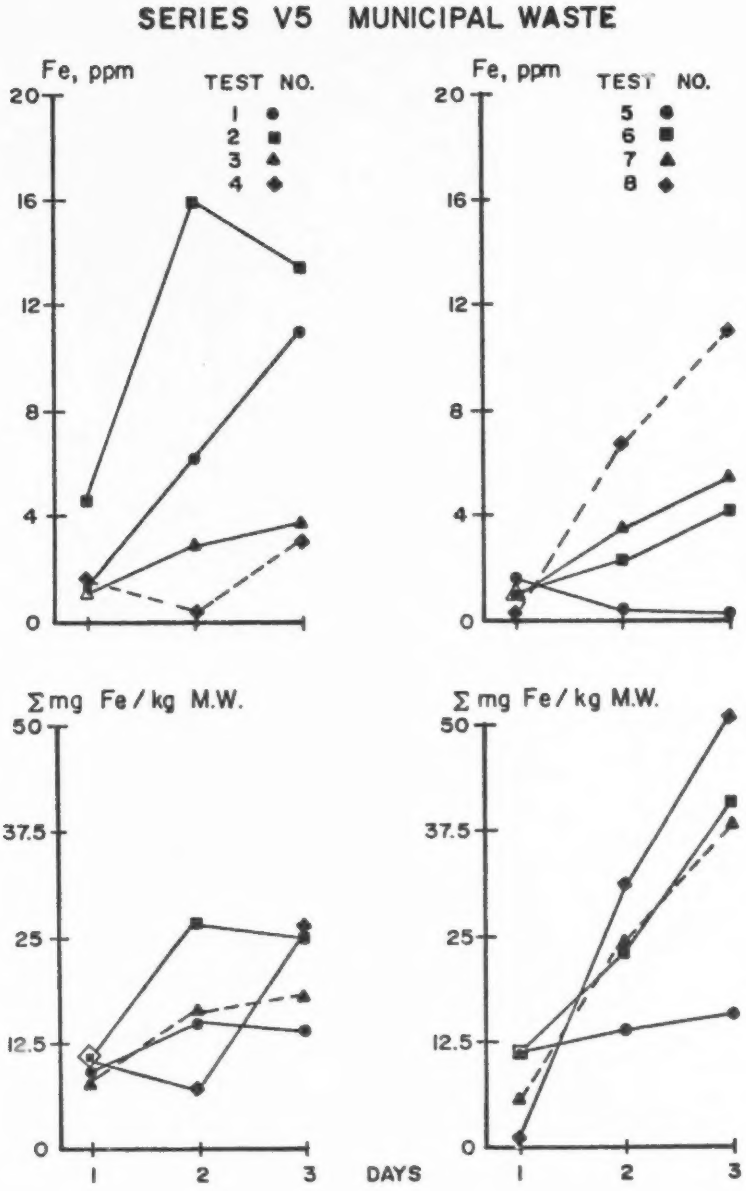

Figure A-113. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Fe.

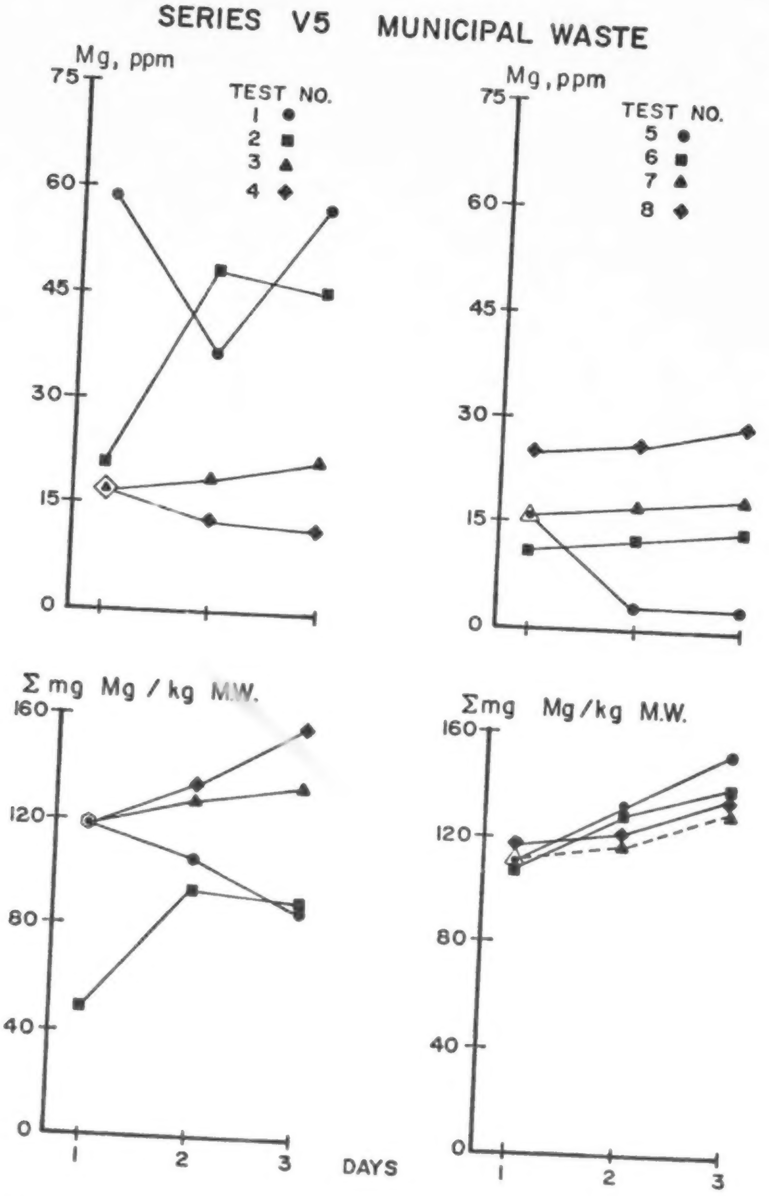

Figure A-114. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Mg.

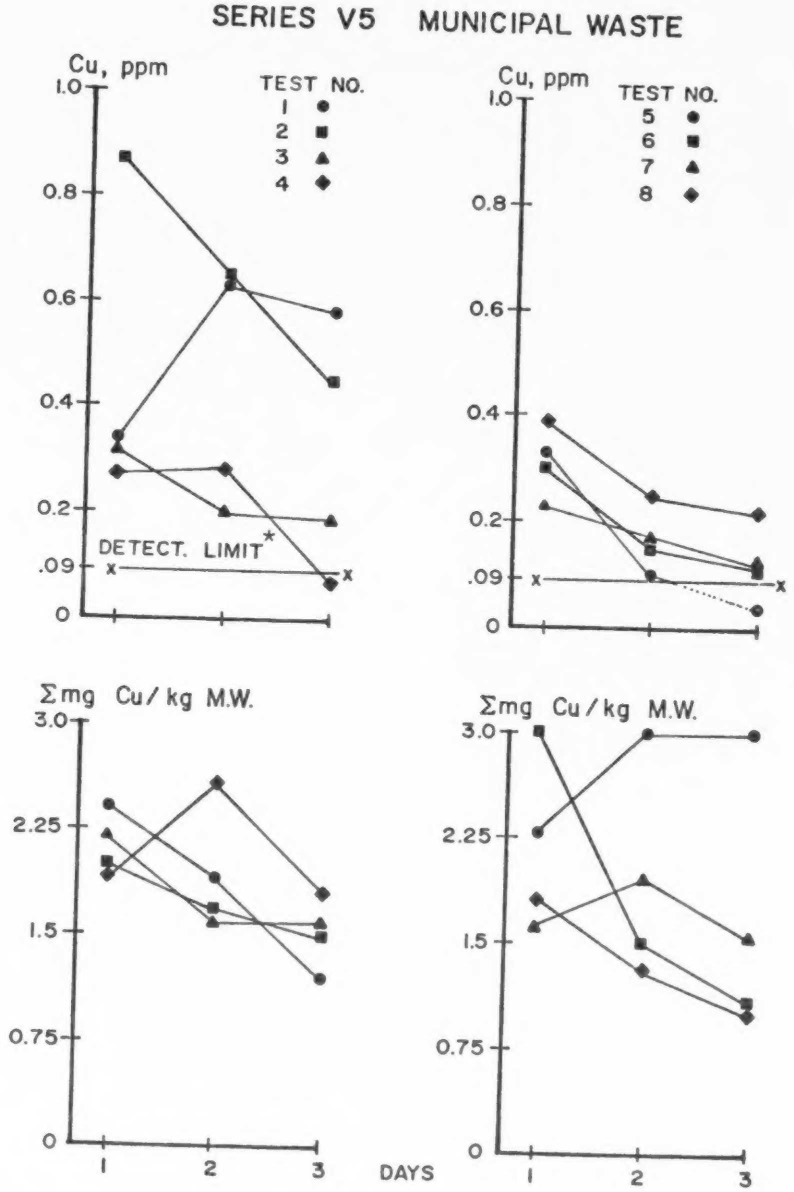

Figure A-115. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Cu.

*Points below detection limit are approximate.

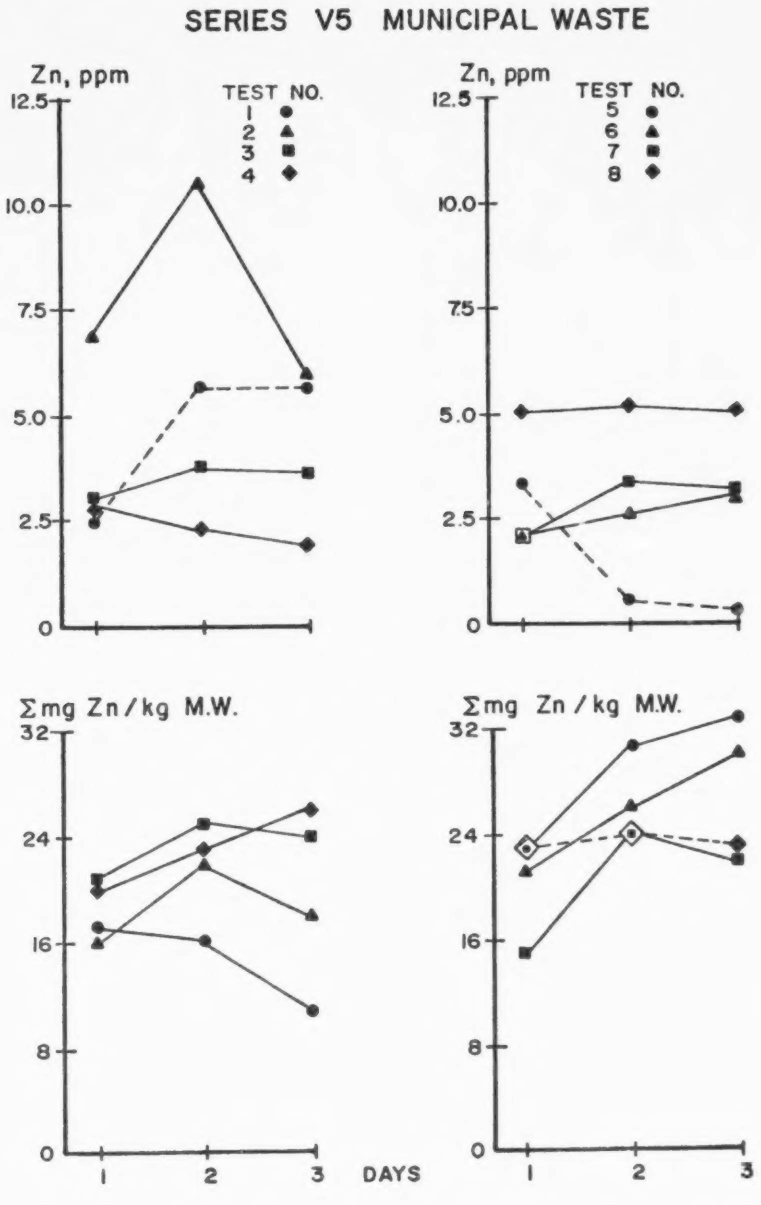

Figure A-116. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Zn.

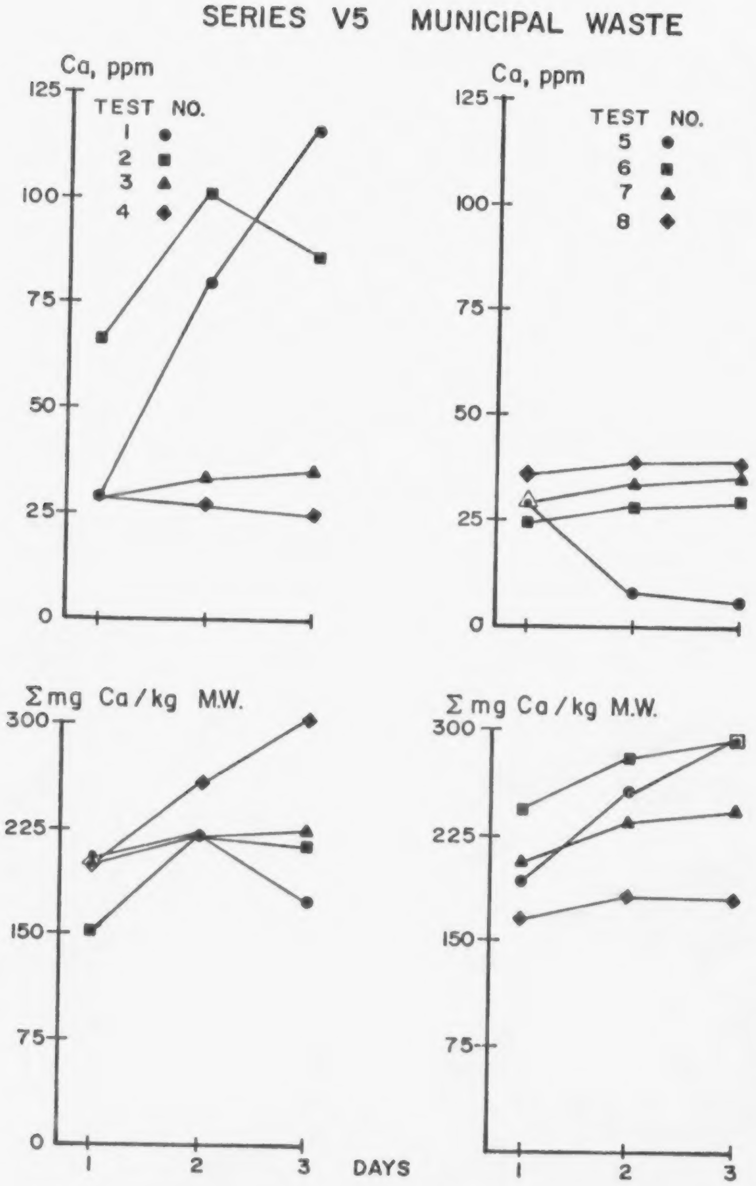

Figure A-117. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. Ca.

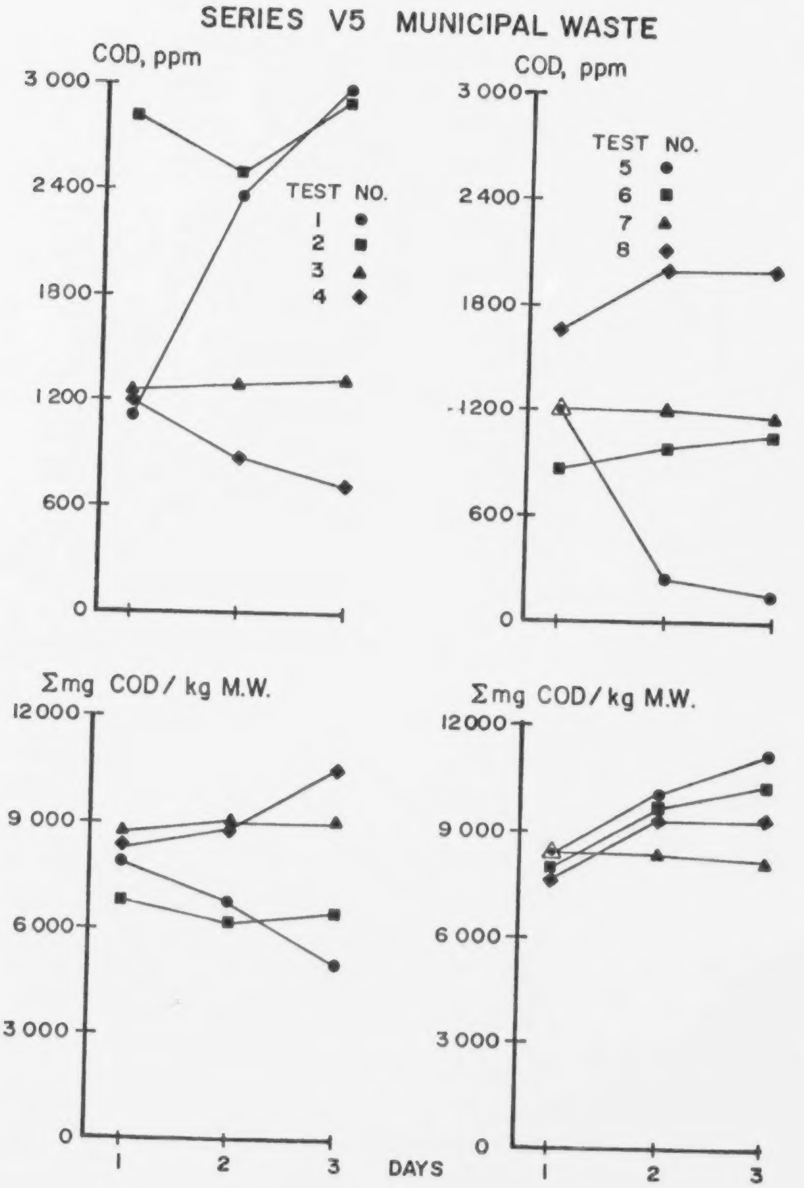

Figure A-118. Test V5 evaluating different contact procedures with shredded municipal solid waste and distilled water. COD.

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